EFFECTS OF ANTHROPOGENIC NUTRIENT ENRICHMENT ON EXOTIC AND RESTORED NATIVE AQUATIC VEGETATION

A Thesis

by

ALLISON PARNELL

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF MARINE RESOURCES MANAGEMENT

May 2011

Major Subject: Marine Resources Management

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Approved by:

Co-Chairs of Committee,	Anna R. Armitage
	Jae-Young Ko
Committee Members,	Georgianne W. Moore
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ABSTRACT

Effects of Anthropogenic Nutrient Enrichment on Exotic and Restored Native Aquatic Vegetation. (May 2011)

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Co-Chairs of Advisory Committee: Dr. Anna R. Armitage
Dr. Jae-Young Ko

Understanding how nutrient input into coastal wetlands influences aquatic vegetation and the fate of anthropogenic nutrient inputs can help improve water quality management plans. The goals of this study were to (1) compare nutrient concentrations in various storage compartments downstream from two point nutrient sources (a sewage treatment plant outfall and a residential detention basin) in Armand Bayou, a coastal brackish wetland in the Galveston Bay (Texas, USA) watershed, and (2) determine if nutrient storage in those compartments helped improve water quality downstream.

Water column nutrients can be assimilated by aquatic vegetation, adsorbed to sediment, or diluted within the system as distance from source input increases. To determine the fate of nutrients to Armand Bayou, I measured nutrient concentrations in the sediment, water column, pore water, and tissue of exotic and restored native plants downstream from a sewage treatment plant and a residential detention basin. To assess nutrient removal potential of a common exotic species, I determined relative growth and nutrient uptake rates of *Eichhornia crassipes*.

Water column total nitrogen, NH_4^+ and NO_3^- concentrations decreased by 95, 96 and 99% downstream from the sewage outfall (~2200 m distance). Water column NH_4^+ and NO_2^- concentrations decreased by 93 and 75% downstream from the detention basin (~2500 m distance). Exotic species *Alternanthera philoxeroides*, *Pistia stratiotes* and *E. crassipes* showed higher aboveground/emergent tissue nutrient content than restored, native *Schoenoplectus californicus* for both tributaries. *Schoenoplectus californicus* had the largest biomass although appeared to be limited in its ability to remove nitrogen from the water column. Nutrient uptake rates by *E. crassipes* were low and did not change with increasing distance from nutrient source, but high relative growth rates in both tributaries suggest the nuisance potential of this exotic species. Low sediment and pore water nutrient concentrations for both tributaries suggest these compartments are not sinks. All plant species did not respond to changes in water column nutrient concentrations with increasing distance from source input suggesting dilution to be the main factor in water column nutrient decline for both tributaries.

This study will provide water quality resource managers guidance on the development of total maximum daily loads (TMDLs) for water bodies impaired by high nutrient loading and the implementation of wetland plants efficient in nutrient removal for water quality improvement.

DEDICATION

I would like to dedicate this work to my parents. Their continuous love and support throughout my life gave me the confidence and determination to try and succeed at the unknown.

ACKNOWLEDGEMENTS

I would like to thank my committee co-chairs, Dr. Anna Armitage and Dr. Jae-Young Ko, and my committee member, Dr. Georgianne Moore, for their guidance and support throughout the course of this research.

Thanks also go to my friends and colleagues, and the department faculty and staff, for making my time at Texas A&M University at Galveston a great experience. I also want to extend my gratitude to Texas SeaGrant, which provided the funding for the research, Mark Kramer with Armand Bayou Nature Center who provided equipment and access to my research sites, Kathy Schwerh and the Laboratory for Oceanographic and Environmental Research who provided the use of equipment for nutrient analysis, the Geochemical and Environmental Research Group who processed all water nutrients, and to my friends and lab mates who provided much needed assistance in the field.

Finally, thanks to my boyfriend, Clif, for his continuous encouragement and tremendous patience. I could not have imagined this entire experience without you.

NOMENCLATURE

ABBA	Armand Bayou Bioassay
ABSS	Armand Bayou Survey Samples
BOD	Biochemical Oxygen Demand
BW	Brookwood Residential Housing Development
CWA	Clean Water Act
EPA	Environmental Protection Agency
GERG	Geochemical and Environmental Research Group
HB	Horsepen Bayou
OC	Organic Content
RGR	Relative Growth Rate
TMDLs	Total Maximum Daily Loads
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids

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1. INTRODUCTION

1.1 Eutrophication of Wetland Habitats

The degradation of estuarine habitats has been influenced by natural and anthropogenic impacts such as subsidence, urban and agricultural development, and the introduction of exotic species, but declines in water quality are of particular concern for recreational and commercial users of estuaries. The decline of water quality within estuarine habitats can be largely attributed to increased nutrient input (Howarth et al. 2002) from point sources such as municipal sewage and industrial waste outfalls, and non-point sources such as urban and agricultural runoff (Davis and Cornwell 1991), resulting in pronounced eutrophication of water bodies (Brix and Schierup 1989).

Eutrophication is a process defined by Nixon (1995) "as an increase in the rate of supply of organic matter to an ecosystem". In freshwater and coastal marine systems, eutrophication can lead to a decline in water column transparency which can decrease the aesthetic value of the system (Smith 2003). Disruption of the natural physical heterogeneity of aquatic systems following channel modifications and vegetation removal (Brix and Schierup 1989) have exacerbated the effects of eutrophication by changing species diversity, productivity and biomass of vascular aquatic plants (Smith 2003). Eutrophication can lead to hypoxic and anoxic conditions by increasing the deposition of organic matter, which then enhances heterotrophic respiration (Diaz and Rosenburg 2008), resulting in oxygen consumption exceeding oxygen supply (NRC 1993). In coastal areas, nutrient enrichment and wastewater effluent are a serious

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This thesis follows the style of *Estuaries and Coasts.*

ecological concern and should be regulated accordingly (NRC 1993).

1.2 Regulation of Water

In the US in 1972, the Federal Water Pollution Control Amendments introduced a permit system for regulating point sources of pollution in an effort to improve water quality in aquatic systems. In 1977 major amendments were enacted to put into effect the Clean Water Act (CWA). Currently, the main objective of water quality management is to control anthropogenic pollution so that water is not degraded beyond its intended uses (Davis and Cornwell 1991). In order to control pollution loads and sources (Elshorbagy et al. 2005), rigorous water quality standards must be developed. The total maximum daily loads (TMDLs) program, which was initiated in the CWA, has become the basis for quality standards in water bodies (Elshorbagy et al. 2005). The purpose of implementing TMDL programs is to determine the maximum allowable pollutant a water body can assimilate without violating water quality standards (Elshorbagy et al. 2005). States, territories and authorized tribes are mandated under Section 303 (d) of the CWA and the U.S. Environmental Protection Agency (EPA) Water Quality Planning and Management Regulations (40 CFR Part 130) to develop TMDLs for water bodies that fail to meet intended uses under technology based controls for pollution (Elshorbagy et al. 2005). Implementation of TMDLs is watershed specific; therefore local field knowledge by resource managers is necessary to develop allowable TMDL standards for any given area.

Strict regulations regarding water quality have led to the use of stormwater retention ponds to manage urban, agricultural and commercial runoff (Fox et al. 2008).

These ponds are primarily designed to lower water column pollutant levels by reducing suspended solids (Mallin et al. 2002) and pollutants associated with sediments such as heavy metals and nutrients (Wong et al. 1999). Aquatic vegetation and soil in stormwater retention ponds help to reduce the volume and rate of stormwater discharge by intercepting and filtering runoff (Lawrence et al. 1997). Therefore, an important component of stormwater management is to limit or prevent the removal of vegetation and changes in soil permeability that can be brought on by increased urbanization (Lawrence et al. 1997).

Nutrient assimilation by aquatic vegetation, adsorption to sediment, or nutrient retention in pore water can decrease water column nutrients downstream from source input (Fig. 1). Wetland systems can provide such nutrient storage compartments that may help maintain water quality in spite of increased nutrient loads. Dilution of water column nutrients can also result in decreased nutrient loads within the system (Fig. 1). A linear decline in water column nutrient concentrations with increasing distance from source input suggests dilution (Officer 1979) (Fig. 2). A curvilinear relationship suggests wetland plants, sediment and pore water are sinks for nutrients (Fig. 2) thus improving water quality downstream more efficiently than expected from dilution alone.

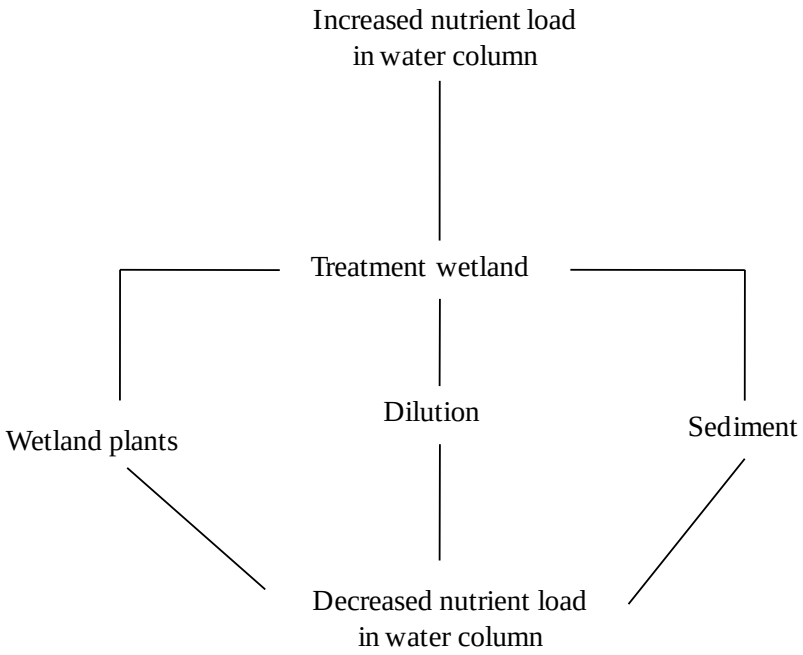

Fig. 1 Conceptual model for nutrient storage compartments in wetlands influenced by increased nutrient loading.

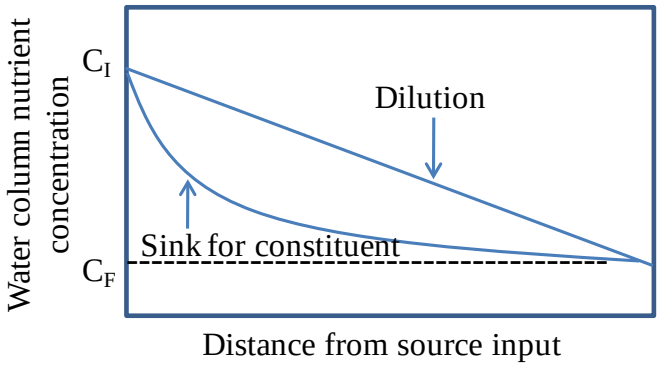

Fig. 2 Conceptual graph illustrating mixing plots (water column nutrient concentration vs. distance from nutrient source) where C_I is the water column nutrient concentration at the source of input and C_F is the water column nutrient concentration downstream from source input. Figure is modified from Officer (1979).

1.3 Phytoremediation of Water Quality

Numerous studies have shown that wetlands can be effective secondary and tertiary processors of wastewater effluent (Kadlec and Wallace 2009). Secondary treatment of wastewater effluent reduces biochemical oxygen demand (BOD) and total suspended solids (TSS) and is the minimum requirement for municipal and industrial effluent before discharge into receiving water bodies (Kadlec and Wallace 2009). Tertiary or advanced treatment goes beyond the limits of secondary treatment and involves complex biochemical processes such as nitrification, denitrification, and phosphorus removal (Kadlec and Wallace 2009). Natural wetlands, from Florida to northwest Canada, have been shown to be effective at removing nitrogen and phosphorus from wastewater effluent as water flows through wetlands (Nichols 1983). Day et al. (2004) reported a reduction in surface water nutrient concentrations from effluent inflow to outflow ranging from 100% for nitrate to 66% for total phosphorus in Louisiana wetlands receiving secondarily treated wastewater. Surface flow wetlands used for water quality management initiated a study on the effects of secondarily treated wastewater discharged into a rich fen in Houghton Lake, Michigan (Mitsch and Gosselink 2007). Researchers found significant reductions in ammonia nitrogen and total dissolved phosphorus as the effluent passed from the point of discharge through the wetlands. Construction of phytoremediation wetlands is increasing due to low costs associated with the development and maintenance of these systems compared to conventional technology-based controls for pollution.

Wetland systems can naturally treat wastewater by dispersing water from a point source over a large area (Boyt et al. 1977), which facilitates nutrient uptake by aquatic vegetation (phytoremediation). In addition to direct nutrient uptake, wetland plants decrease water flow, which increases settlement of solid particulates and volatilization rates and enhances microbial activity, removing organic and inorganic pollutants from aquatic habitats (Dhote and Dixit 2009). However, more land is required to sustain these systems and vegetation uptake of excess nutrients may decrease during the winter in temperate regions (Brix and Schierup 1989).

Macroscopic aquatic plants growing in wetlands (macrophytes) are potentially useful for nutrient uptake from the water column. In systems influenced by wastewater, macrophyte species with rapid growth and high plant tissue nutrient content tend to reflect high rates of nutrient uptake from the water column (Reddy and DeBusk 1987). Additionally increased biomass can influence the nutrient storage capacity in plants (Reddy and DeBusk 1987). Therefore wetland plant species with high growth rates, high plant tissue nutrient content and high biomass accumulation would yield higher potential for nutrient removal from the water column. In the US, studies have reported *Schoenoplectus californicus* (California bulrush) as a preferred choice to use in removing excess nutrients from constructed wetlands because of its high tolerance to elevated ammonia levels (160 to 170 mg/L) (Surrency 1993). In a study of nutrient removal by constructed and natural wetlands in the Las Vegas Valley, Nevada, *S. californicus* showed the highest nutrient content compared to other bulrush species (Adhikari et al. 2010). In addition, low nitrogen input coupled with low water and

sediment nitrogen concentrations, nitrogen removal via plant assimilation was high in a constructed wastewater treatment wetland dominated by *S. californicus* and other bulrush species. *Schoenoplectus californicus* has also been planted in constructed wetlands in New Zealand for wastewater treatment and reported to exhibit higher live biomass and shoot density in constructed wetlands compared to natural systems (de Lange et al. 1998). *Schoenoplectus californicus* in constructed wetlands had higher aboveground tissue total nitrogen and phosphorus content (1.93 and 0.36% dry weight, respectively) with a gravel substrate compared to natural wetlands with silt (1.15 and 0.22% dry weight, respectively) and sand substrates (1.13 and 0.14% dry weight, respectively) (de Lange et al. 1998). Higher nutrient content of this species in constructed wetlands impacted by wastewater is probably a result of a combination of high nutrient loading and high biomass accumulation.

There are a variety of exotic macrophytes that remove nutrients from coastal waterways. *Alternanthera philoxeroides* (alligatorweed), an emergent exotic species, showed lower nitrogen and phosphorus content (0.58 and 0.25% dry weight, respectively) compared to a floating species, *Eichhornia crassipes* (water hyacinth), (1.02 and 0.40% dry weight) in response to sewage effluent (Scarsbrook and Davis 1971). Nitrogen removal for *A. philoxeroides* at the end of the experiment was less than when the plant was originally sampled although slightly higher for phosphorus removal in the sewage effluent (Scarsbrook and Davis 1971). However, higher nitrogen and phosphorus content (2.87 and 0.32% dry weight, respectively) for *A. philoxeroides* as well as for *E. crassipes* (2.64% N and 0.43% P dry weight) were reported by Boyd

(1969) in a study of aquatic plants for nutrient removal of polluted waters. This contrasts what Scarsbrook and Davis (1971) reported but still provides insight on the potential of nutrient removal of these aquatic macrophytes. Boyd (1969) also states wetland plants that exhibit high density should be capable of lowering nutrient levels.

Floating, exotic macrophyte *E. crassipes* has been intensely studied for its capacity to assimilate nutrients in enriched systems (Cornwell et al. 1977; Dunigan et al. 1975; Rogers and Davis 1972; Sheffield 1967). The ability of this species to exhibit substantial vertical growth allows *E. crassipes* to increase its growth potential (Reddy and DeBusk 1987) which aids in nutrient removal from the water column. *Eichhornia crassipes* showed a positive response to sewage effluent and increased in plant total nitrogen and phosphorus with the highest content in October (1.02 and 0.40% dry weight, respectively) (Scarsbrook and Davis 1971). Nutrient removal at the end of the experiment resulted in *E. crassipes* removing 6.93 g of nitrogen and 2.87 g of phosphorus from sewage effluent (Scarsbrook and Davis 1971). During summer months in nutrient enriched microcosms *E. crassipes'* nitrogen and phosphorus content (3.05 and 0.58% dry weight, respectively) (Reddy and DeBusk 1985) was higher compared to the values reported by Scarsbrook and Davis (1971) (1.02 and 0.40% dry weight, respectively) for this species in sewage effluent. *Eichhornia crassipes* also showed the highest removal of phosphorus (93%) during the summer which coincided with the rapid decline in water column phosphorus (Reddy and DeBusk 1985). However, higher nitrogen and phosphorus content (3.53 and 0.68% dry weight, respectively) was reported in this species during winter (Reddy and DeBusk 1985) which was probably attributed to

slow growth and luxury uptake (Reddy et al. 1983). This species has also been assessed for phytoremediation of nutrients in urban retention ponds (Fox et al. 2008). Fox et al. (2008) showed nitrogen removal from the water column significantly increased as *E. crassipes* plant tissue nitrogen increased which also coincided with increased plant biomass.

In addition to *E. crassipes* demonstrating high nutrient removal potential, nutrient uptake rates have been widely investigated to determine how much nutrients this species is capable of removing from enriched environments (Brix 1997; DeBusk et al. 1995; Imaoka and Teranishi 1988; Reddy and Tucker 1983). In constructed wetlands Brix (1997) demonstrated *E. crassipes* had higher uptake capacities (~0.548 g N m^{-2} d^{-1} and 0.096 g P m^{-2} d^{-1}) than emergent macrophytes which can result in higher plant tissue nutrient content or biomass. DeBusk et al. (1995) calculated maximum assimilation rates (up to 0.777 g N m^{-2} d^{-1} and 0.2 g P m^{-2} d^{-1}) during the summer for this species under nutrient enriched conditions. Due to this species' rapid growth rate, the nutrient uptake rate (based on growth rate and nutrient content) was higher than that of another floating species, *Lemna obscura* (duckweed) (DeBusk et al. 1995).

Pistia stratiotes (water lettuce) is another example of a floating, exotic macrophyte that has been studied for its treatment potential in sewage effluent (Zimmels et al. 2006). *Pistia stratiotes* removed 67% of phosphorus from nutrient enriched waters and showed higher phosphorus and nitrogen content (0.80 and 3.62% dry weight) compared to *E. crassipes* (0.58 and 3.05% dry weight) during summer months (Reddy and DeBusk 1985). In contrast, nitrogen and phosphorus content were 1.5 times higher

in *E. crassipes* (2.15 and 1.67% dry weight) than *P. stratiotes* (1.65 and 1.03% dry weight) grown under enriched conditions (Aoi and Hayashi 1996). Similar comparisons were found in mixed cultures of *E. crassipes* and *P. stratiotes* grown under increased nutrient levels (N: 3.93 and 2.83; P: 0.71 and 0.70% dry weight, respectively) (Agami and Reddy 1990). High plant tissue nutrient content helped explain the increase (by 3 fold) in both species' biomass that resulted in the complete cover of the water surface.

Measuring plant tissue nutrient content of various wetland plants downstream from sources of enriched nutrient input as well as relating relative uptake rates of plants to water column nutrient concentrations can quantify nutrient retention within the system. I would expect wetland plants efficient in nutrient retention to yield increased plant biomass and nutrient uptake rates. This can help determine if specific aquatic wetland plants are effectively serving as nutrient storage compartments that will ultimately improve water quality downstream.

1.4 Nutrient Retention in Pore Water

Pore water can provide an effective nutrient storage compartment in eutrophic water bodies. Pore water phosphate (PO_4-P) showed high concentrations with the highest concentrations (0.91 mg/L P) sampled during the summer in shallow eutrophic areas of the Florida Everglades as a result of external nutrient loading (Vaithiyanathan and Richardson 1997). Pore water ammonium (NH_4-N) concentrations were one to two orders of magnitude higher than overlying surface waters with the highest concentrations (18 mg/L N) measured in eutrophic sites during the summer. In a subtropical marsh ecosystem pore water ammonium and phosphate concentrations were higher compared

to the overlying water column indicating these dissolved ions are being produced in the sediment (Soto-Jimenez et al. 2003). Pore water nitrite (NO_2^-) concentrations were highest between the sediment-water interface to a depth of 6 cm as a result of nitrate reduction (Soto-Jimenez et al. 2003).

1.5 Nutrient Retention in Sediment

Sediment is an important storage compartment for phosphorus. Phosphorus adsorbs to sediment in a process of rapid exchanges between porewater and sediment; binding to the surface of a sediment particle (Dunne and Reddy 2005). Organic forms of phosphorus readily sorb to sediments composed of clay and organic matter (Dunne and Reddy 2005), effectively immobilizing the nutrient (Sakadevan and Bavor 1998). In constructed wetlands receiving secondarily treated wastewater, sediment composed of native silty clay loam sediment showed a relatively high capacity for phosphorus uptake from solution and surface sediments indicated higher sorption potentials than subsediment layers (Geiger et al. 1993). This supports Sakadevan and Bavor's (1999) study where they reported an increase in sediment total phosphorus and attributed this increase to adsorption of phosphate. However, varying degrees of nutrient input can alter phosphorus adsorption capacity to sediment (Sakadevan and Bavor 1998). They found low phosphorus adsorption (30%) with high inputs of total phosphorus (>200 mg/L P) and more than 50% adsorption of phosphorus with low inputs of total phosphorus (0 to 200 mg/L P). In the Florida Everglades, Vaithiyanathan and Richardson (1997) detected higher sediment total phosphorus content, as a result of

adsoprtion processes, at nutrient enriched sites compared to concentrations downstream (8-10 km) from the enriched sites.

Sediment is also an important sink for nitrogen. In wetland ecoystems worldwide the largest reservoir of total nitrogen is sediments. Most sediment nitrogen is in organic forms (100 to 1000 g N m^{-2}). Sediment inorganic nitr ogen content are two orders of magnitude less than organic content (Bowden 1987) because inorganic oxidized forms of nitrogen, such as nitrate (NO_3^-) and nitrite (NO_2^-), are unable to adhere to solid substrates but ionized ammonium (NH_4^+) is capable of sorption to organic and inorganic substrates (Kadlec and Wallace 2009). It is within the reduced layer of sediments that NH_4^+ becomes stable and adsorbed to sediment or taken up by plants and microbes (Faulkner and Richardson 1989). Therefore NH_4^+ may be removed from the water column through detrial and inorganic sediment exchange (Kadlec and Wallace 2009) subsequently providing a sink for nutrients and helping to improve water quality in impaired waterways. In the Netherlands a survey was conducted to determine the efficacy of constructed littoral wetlands for water quality improvement (Sollie et al. 2008). Sediment total nitrogen tended to be higher, although not significant, in the constructed sites compared to the natural sites for both vegetated and bare plots, suggesting that this wetland compartment provides a nitrogen sink and can help to improve water quality within constructed wetlands. In the aforementioned Everglades study, sediment nitrogen content remained constant downstream from nutrient enriched sites and overall, exhibited relatively high content (2.5-3.5%) (Vaithiyanathan and Richardson 1997) as a result of low denitrification potential in the peat soil (Gordon et

al. 1986) and the presence of nitrogen fixing algae (Vaithiyanathan and Richardson 1997).

1.6 “Big Picture”

Urbanized watersheds in coastal areas are characterized by anthropogenic impacts involving point sources of pollution. For example, by discharging secondarily treated wastewater into a restored wetland rather than directly into a flowing water body, there is potential for more nutrients to be removed from the water column by wetland plants and stored in sediments. A residential detention basin designed to provide flood storage also provides potential for more nutrient removal from the water column before being discharged into adjacent wetlands, by increasing the amount of time runoff is in contact with aquatic vegetation.

The focus of this study was to investigate how effective restored wetlands and detention basins are at lowering nutrient loads in coastal waterways. In particular, I sought to determine if water flow through restored wetlands and detention basins in Armand Bayou (near Houston, Texas) improved water quality downstream from sources of nutrient input. Comparison of the relative amount of nutrients stored in the plants, sediment, and water column of the wetland ecosystem helped determine the fate of excess nutrients and the removal potential of wetland plants and sediment.

1.7 Study Objectives and Hypotheses

The development of TMDL standards for an urbanized restored wetland such as Armand Bayou requires an understanding of how nutrient input will influence aquatic vegetation and how restored wetlands might help improve water quality. The purpose of this study

was to compare nutrient concentrations in various storage compartments downstream from two point sources in Armand Bayou and determine if the storage compartments contribute to improved water quality downstream. Nutrients discharged from a sewage treatment plant into the water column are expected to decrease as a result of direct plant uptake, adsorption to sediment or retention in pore water or become diluted within the system as distance from nutrient source input increases (Fig. 1). Due to increased retention time in a detention basin, water column nutrient concentrations are expected to further decrease when discharged into adjacent wetlands as a result of direct plant uptake, adsorption to sediment or retention in pore water or become diluted within the system as distance from nutrient source input increases (Fig. 1). Increased distance from nutrient source input in relation to water column nutrient concentrations was used to determine the fate of these constituents within the system. Dilution of water column nutrients will be indicated by a negative linear relationship between water column nutrient concentration and increased distance from nutrient input (Fig. 2). A negative curvilinear relationship will suggest a sink for nutrients via assimilation by aquatic plants, sediment adsorption or pore water retention (Fig. 2).

Objective 1: Compare nutrient concentrations in various storage compartments between two point sources in Armand Bayou.

Hypothesis 1: Nutrient concentrations in the water column, pore water, sediment, and plants will be higher for the sewage treatment plant compared to the storage compartments for the residential detention basin.

Objective 2: Assess if water column nutrient concentrations decrease downstream from sources of input and if nutrient uptake by aquatic vegetation, nutrient adsorption to sediment and nutrient retention in pore water improves water quality within Armand Bayou.

Hypothesis 2: Water column nutrient concentrations will decline downstream from sources of input indicating improved water quality; plant tissue, sediment and pore water will have the highest nutrient concentration near nutrient input sources, suggesting that they are serving as nutrient storage compartments.

Objective 3: Compare nutrient retention capacities between exotic and restored, native aquatic vegetation species.

Hypothesis 3: Exotic aquatic vegetation will have a higher nutrient retention capacity than restored, native aquatic vegetation.

Objective 4: Assess potential for a common exotic species, *Eichhornia crassipes*, to take up nutrients from the water column by determining nutrient uptake rates.

Hypothesis 4: Rates of nutrient uptake and relative growth for *E. crassipes* will be highest near the sewage treatment plant and detention basin due to increased nutrient input.

2. METHODS

2.1 Site Selection

Armand Bayou was chosen as a model system to quantify the effectiveness of restored wetlands and detention basins influenced by anthropogenic nutrient input in coastal waterways. Armand Bayou is a small coastal watershed (approximately 155 km^2) located west of Galveston Bay and approximately 32 km southeast of Houston in Harris County, Texas. The watershed has been subjected to increased urbanization as it encompasses the cities of Pasadena and Clear Lake and the National Aeronautics and Space Administration (NASA), Johnson Space Center, Ellington Air Field and the Bayport petrochemical complex (East and Hogan 2003). It is characteristic of a eutrophic waterway because high levels of total phosphate, orthophosphate, ammonia and nitrate are present in the water (McFarlane 1991). Armand Bayou includes a major tributary, Horsepen Bayou (East and Hogan 2003). Horsepen Bayou is impacted by secondarily treated wastewater effluent discharged from the Robert T. Savely water reclamation facility (Fig. 3). A large residential housing development, Brookwood, is adjacent to Armand Bayou and collects stormwater run-off in a detention basin that connects to the bayou via a culvert (Fig. 3). In Harris County some detention basins are engineered to permanently hold water and are referred to as wet detention basins (Harris County Flood Control District 2010). Wet detention basins like the one in Brookwood can improve water quality by increasing sedimentation rates and housing vegetation that can take up excess nutrients from the water column (Walker 1987).

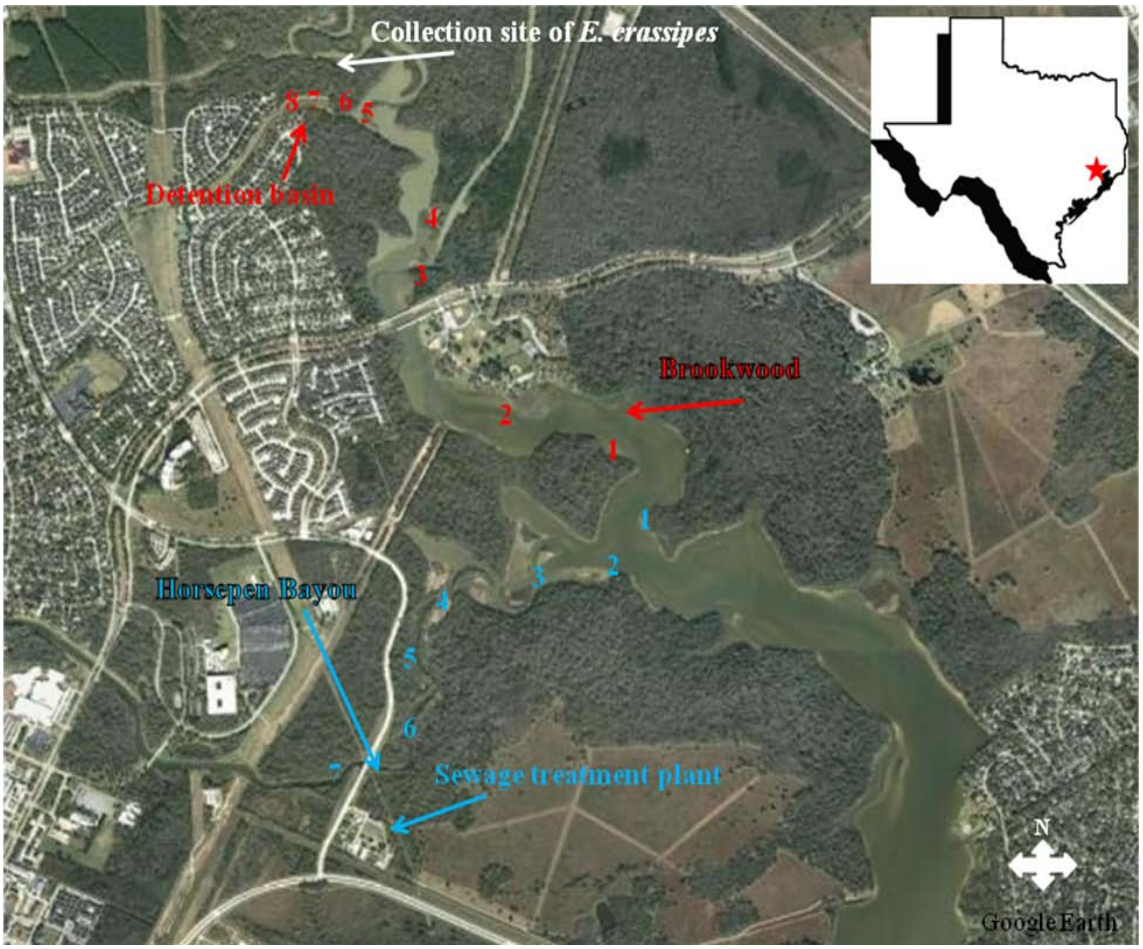

Fig. 3 Transect stations for Horsepen Bayou and Brookwood in Armand Bayou in Harris County, Texas. Upstream from the detention basin was the 2009 collection site of *E. crassipes* for the bioassay experiment (accessed from Google Earth).

2.2 Armand Bayou Survey Samples (ABSS)

Armand Bayou contained 111 hectares of emergent tidal freshwater and brackish marshes as recently as 1956 (McFarlane 1991), but rapid subsidence in the 1980s caused an almost complete loss (90% reduction) of the marshes. Restoration efforts began in the early 1990s. Early restoration projects focused on establishing *Spartina alterniflora* (smooth cordgrass) along the banks of the lower reaches of the bayou with summer salinity ranges of ~8-14‰ (Kramer 2010). Later, upstream restoration work in lower salinities (0-6‰) and open water areas (~1 m depth) focused on the native species *Schoenoplectus californicus* (California bulrush).

Several exotic species have been introduced to Armand Bayou; including *Alternanthera philoxeroides* (alligatorweed), *Eichhornia crassipes* (water hyacinth) and *Pistia stratiotes* (water lettuce). Each of these species can proliferate in high temperature and low salinity conditions; subsequently impeding waterways by forming thick, dense mats on the water surface. Large infestations of such species increase organic matter accumulation, subsequently increasing sedimentation rates, accelerating eutrophication, and reducing water depth (Charudattan 2001). Additionally, invasive exotics can decrease local biodiversity by outcompeting native flora and their associated native fauna (Charudattan 2001). *Eichhornia crassipes* is considered a pest plant species (Dunigan et al. 1975) and thrives well in tropical and subtropical areas worldwide. In Armand Bayou, resource managers currently are using chemical controls to help eradicate *E. crassipes* because of its rapid growth and reproduction (Dunigan et al. 1975). *Alternanthera philoxeroides* is a summer perennial weed that reproduces rapidly

through stem fragments (Shen et al. 2005) and has a high tolerance to herbicides (Maddox et al. 1971). Managers at Armand Bayou are utilizing biological controls, i.e., alligatorweed flea beetle (*Agasicles hygrophila*) to limit *A. philoxeroides* coverage. *Pistia stratiotes,* a tropical/subtropical perennial species, (Boyd 1970) is known to reproduce rapidly in a nutrient enriched environment (Zimmels et al. 2006). This species is currently not being eradicated by chemical or biological means in Armand Bayou; however, resource managers are relying on salinity levels to control this pest species (Kramer 2010). The spread of these exotic species may be facilitated by high nutrient input to Armand Bayou, subsequently increasing management costs while lowering the ecological and aesthetic value of this waterway.

In June 2009, transects were established along two tributaries in Armand Bayou. The first led from the wet detention basin in Brookwood (hereafter referred to as Brookwood or BW) to the opening of Horsepen Bayou (~2500 m distance). The second transect led from the sewage treatment plant in Horsepen Bayou (hereafter referred to as Horsepen or HB) to the Brookwood transect in the main Armand Bayou tributary (~2200 m distance). The Horsepen transect had seven stations; all stations were approximately 300 to 400 m apart (Fig. 3). The Brookwood transect had eight stations; most stations were approximately 100 to 400 m apart with stations BW5 to BW4 and BW3 to BW2 more than 600 m apart (Fig. 3). The Horsepen transect station HB7 (~160 m) was located closest to the sewage outfall and the Brookwood transect stations BW7 and BW8 were located on opposite ends of the detention basin. The salinity at the upstream end of both transects was 0‰. As distance from the nutrient source increased, there was a

slight increase in salinity, but salinity did not exceed 6‰. Stations were established where at least one of three aquatic plant species was present: *S. californicus*, *A. philoxeroides*, or *P. stratiotes*. *Schoenoplectus californicus* is a restored, native, rooted emergent species that has been the focus of recent restoration efforts. *Alternanthera philoxeroides* is an exotic, rooted emergent species and *P. stratiotes* is an exotic, floating species. These exotic species were chosen because they are often prolific in summer months.

In June 2009 I performed a series of surveys along both tributaries to determine plant abundance, biomass and aboveground and belowground plant tissue nutrient content of restored, native *S. californicus* and exotic *A. philoxeroides* and emergent and submerged plant tissue nutrient content of exotic *P. stratiotes* (*E. crassipes* was extremely rare in early summer 2009 and was therefore not included in these surveys). I also determined water column and pore water nutrient concentrations and sediment nutrient and organic content and grain size.

2.2.1 Stem Density

Replicates (n = 3 per species) of stem density were measured for *S. californicus* and *A. philoxeroides*. The number of stems of each species at each station was counted within a haphazardly placed 20 cm x 10 cm quadrat. Station area (m^2) was visually estimated based on the discrete boundaries of *S. californicus* and *A. philoxeroides*. Total plant density was calculated by dividing stem density by area of the station (# of stems/station). Stem density for *P. stratiotes* was determined based on the number of plants present at each station and divided by the station area (# of plants/station). The

reason for this exception was due to the extremely low abundance of this species at all stations.

2.2.2 Plant Biomass

In order to estimate the total amount of nutrients stored in the aboveground/emergent and belowground/submerged plant compartment of each species, total plant biomass was estimated as the dry weight of the entire plant (kg dry weight/station). Replicates (n = 3 per species) of aboveground and belowground biomass for *S. californicus* and *A. philoxeroides* were measured using a modified method from Hoagland et al. (2001). Biomass was estimated within three 20 cm x 10 cm plots for *S. californicus* and *A. philoxeroides*. Within each plot, *S. californicus* and *A. philoxeroides* were collected by carefully clipping aboveground plant tissue at the water-atmosphere interface. Remaining aboveground biomass in the water column was separated from belowground biomass back in the lab. Belowground biomass of *S. californicus* and *A. philoxeroides* were sub-sampled using a 7 cm diameter aluminum sediment corer. The corer was placed over the remaining aboveground plant tissue and inserted into the sediment to a depth of 20 cm. Insufficient collection of *A. philoxeroides* belowground plant biomass from most Horsepen stations resulted in the determination of total plant biomass for only two stations. Due to the extremely low abundance of *P. stratiotes*, biomass was estimated based on the total number of plants at each station. *Pista stratiotes* was removed as a whole specimen to ensure that no fragments or loose plants were released downstream. All samples were frozen prior to processing. All cores were rinsed through a 1 mm sieve to separate the remaining aboveground tissue and root material.

Plant material was thoroughly rinsed with de-ionized water to remove adhering sediment and debris. Each species was separated into live and dead plant material. All live stems/leaves were separated from the roots. All plant material was oven dried at 70°C and weighed.

2.2.3 Plant Tissue Nutrient Content

Replicates (n = 3 per species) of aboveground and belowground plant tissue for *S. californicus* and *A. philoxeroides,* and emergent and submerged plant tissue for *P. stratiotes* were collected for nutrient analysis. Insufficient collection of *A. philoxeroides* belowground plant biomass from most Horsepen stations resulted in the determination of the nitrogen content for only two stations and the phosphorus content for one station. All plant material was thoroughly rinsed with de-ionized water to remove adhering sediment and debris. *Pistia stratiotes* was separated into emergent (stems/leaves) and submerged (roots) components. All plant components were oven dried at 70°C. Once dry, plant samples were ground into a fine powder with a Thomas Wiley Mini-mill. All samples were contained in plastic scintillation vials and capped tightly to keep atmospheric moisture out of samples. Plant tissue was analyzed for total phosphorus by dry oxidation acid hydrolysis extraction of two (17-20 mg) subsamples of each sample followed by colorimetric analysis of phosphate content of the extract (Fourqurean et al. 1992). Colorimetric analysis was conducted on a Shimadzu UV spectrophotometer UV-1800 at 885 nm. Carbon and nitrogen content was determined using a PerkinElmer 2400 CHNS/O analyzer.

2.2.4 Water Column and Pore Water Nutrient Concentration

Replicates (n = 3 per station) of water column samples and one pore water sample were collected at each station. Water column samples were collected in 1 L dark, plastic bottles and stored on ice. In the lab, the water was decanted and filtered through Whatman GF/F glass microfiber filter paper and frozen prior to processing. To collect pore water, a PVC pipe (35 cm in length) with attached t-joint (2 cm diameter) was inserted into the top 10 cm of sediment and moved back and forth three times to collect saturated sediment. The saturated sediment was stored in a 60 ml Falcon tube on ice. In the lab, the Falcon tubes were centrifuged for 15 minutes to separate the pore water from the sediment. The water was decanted and filtered through Whatman GF/F glass microfiber filter paper and frozen prior to processing. All samples were processed by the Geochemical and Environmental Research Group (GERG) at Texas A&M University under QA/QC guidelines. Nitrate and nitrite analyses were based on the methodology of Armstrong et al. (1967) and utilized a ground Cd column for reduction of NO_3^- to NO_2^-. Orthophosphate was measured using chemistry based on the investigations of Bernhardt and Wilhelms (1967) with the modification of hydrazine as reductant. Ammonium analysis was based on the method of Harwood and Kuhn (1970). The autoanalyzer method was modeled after those developed and commonly used for seawater analyses (Strickland and Parsons 1972). Total nitrogen analysis was based on the utilization of potassium persulfate ($K_2S_2O_8$) with pressure and heating to accomplish the complete decomposition and oxidation of elemental components in organic matter to a detectable form. Total nitrogen was determined in the current configuration of the

Technicon II AutoAnalyzer. The autoanalyzer method was modeled after those developed and commonly used for seawater analyses (Valderrama 1981). Due to digestion problems, total phosphorus was unattainable.

2.2.5 Sediment Nutrient Content

Sediment samples (n = 3 per station) were collected using a Van Veen grab (21 cm x 20 cm). Samples were frozen prior to processing. Sediment samples were oven dried at 70°C and ground into a fine powder with a mortar and pestle. Carbon, nitrogen and phosphorus content were determined using the same methods as plant tissue analysis.

2.2.6 Sediment Organic Content and Grain Size Analysis

Organic content and grain size analysis (n = 3 per station) was measured in a sub-sample of the soil collected in the Van Veen grab. Sediment organic content was determined by loss of mass during ignition at 500°C for eight hours. The difference in the dry weight of the sample before and after burning determined the amount of organic content present in the sample. Sediment particle size analysis was determined using a modified hydrometer method of Bouyoucos (1962). Sodium metaphosphate was added to each sample and placed on a shaker table for 24 hours to break up clumps within the sample. Temperature (between 24.4° and 15.6°C) and hydrometer readings were recorded for each sample at 40 seconds and 2 hours (±2 minutes) after the start time. The corrected hydrometer reading at the end of 40 seconds was divided by the dry weight of the soil and multiplied by 100 and represents the percentage of silt plus clay. This percentage (silt + clay) was subtracted from 100 and represents all the sand in the sediment. The corrected hydrometer reading at the end of 2 hours was also divided by the dry weight of

the soil and multiplied by 100 and represents percent clay. The percentage of silt was obtained by the difference between percent clay and percent silt + clay.

2.2.7 Statistical Analyses

Least squares linear regression analyses and curve estimations were used in SPSS to determine if there was a relationship between distance from nutrient source (independent variable) and all dependent variables (aboveground/emergent and belowground/submerged plant tissue nutrient content, stem density, plant biomass, sediment nutrient and organic content, and water column and pore water nutrient concentrations). I ran both linear and curvilinear regressions and reported the line with the best fit (with the highest R^2 value). If the relationship was linear, then water column nutrients were diluted; if the relationship showed a negative exponential decrease, then water column nutrients were being assimilated by plants, adsorbed to sediment or retained in pore water (Fig. 2). Statistical analyses were done separately for each plant species and tributary. *Alternanthera philoxeroides* was not included in statistical analyses due to the fact that this species was found at only a few stations in both tributaries.

2.3 Armand Bayou Bioassay (ABBA)

To estimate nutrient availability and potential nutrient uptake rates by exotic vegetation, I conducted a bioassay experiment using exotic species *Eichhornia crassipes*, where nutrient-starved plants were placed in the tributaries for one week. *Eichhornia crassipes* was chosen over *P. stratiotes* in this experiment because of its larger biomass and its more robust structure allowing for easier collection, transportation and deployment

(Texas Parks and Wildlife Department exotic species permit no. RES 05 09-096). After collection, nutrient contents in the plant tissue revealed nutrient availability over several days, in contrast to the collection of water samples which only reveal nutrient contents as a "snapshot" in time. Determining the nutrient retention capacity and calculating nutrient uptake rates of this species helped to quantify nutrient retention time in the system.

2.3.1 *E. crassipes* Plant Tissue Nutrient Content

Eichhornia crassipes was collected upstream from the detention basin (Fig. 3) in an area not affected by either source of nutrient input. Live, whole specimens of *E. crassipes* were carefully collected from the field to ensure that no fragmentation occurred and that no free plants floated downstream. All plants were placed in 37 L glass aquarium tanks with aerators. De-ionized water was placed in the tanks to create a nutrient-poor environment. Plants remained in the tanks for 7 days to ensure that tissue nutrient contents were lowered to a constant level. Plants were centrifuged for 60 seconds using a low velocity centrifuge and weighed to achieve a standard initial wet weight. Individual plants (27.690 to 47.437 g wet weight) were placed in an enclosure (approximate dimensions 30 x 30 cm) made of bird netting (mesh size = 1 cm^2) with four attached floats. The enclosed plants were transported to the field inside aerated coolers. Three enclosures were deployed 2 meters apart at each of the 15 stations, with a weight attached by a polypropylene rope to keep the enclosures in position. After seven days, I removed each sample from the enclosure and stored it in a plastic bag to prevent contamination and transported all samples back to the laboratory in a cooler. All plants

were thoroughly rinsed with de-ionized water to remove adhering debris and spun in a low velocity centrifuge to attain a standard wet weight. The plants were sorted into emergent (stem/leaves) and submerged (roots) components and oven dried at 70°C. Once dry, plant samples were grounded into a fine powder with a Thomas Wiley Mini-mill. All samples were contained in plastic scintillation vials and capped tightly to keep atmospheric moisture out of samples. Plant tissue was analyzed for carbon, nitrogen and phosphorus using the same methods as described previously.

2.3.2 *E. crassipes* Plant Nutrient Uptake Rate and Relative Growth Rate

I calculated *E. crassipes* nutrient uptake rates in order to quantify nutrient retention time (how much nutrients are removed from the system over a given time) and nutrient storage capacity of this exotic species in Armand Bayou. Nutrient uptake rates for emergent and submerged plant tissue in *E. crassipes* were calculated individually using Eq. 1 and added to get total plant uptake rate:

$$\frac{[(\Delta \text{ dry weight g } \textit{E. crassipes}) \text{ x } (\text{N\% dry weight } \textit{E. crassipes})]}{\text{Time (days)}} = \text{g N d}^{-1}\ \textit{E. crassipes} \qquad (1)$$

Plant relative growth rate (RGR) was determined from the equation of Mitchell and Tur (1975) as:

$$RGR = (\ln_{x2} - \ln_{x1}) / (T_2 - T_1) \qquad (2)$$

where x_1 and x_2 are the dry mass (g) at times T_1 and T_2, respectively.

2.3.3 Statistical Analyses

A least squares linear regression analysis and curve estimation were used in SPSS to determine if there were relationships between *E. crassipes* emergent and submerged

plant tissue nutrient content, plant relative growth rate, and nutrient uptake rate (dependent variables) and distance from the nutrient source (independent variable). Statistical analyses were done separately for each tributary.

3. RESULTS

3.1 Armand Bayou Survey Samples (ABSS)

3.1.1 Water Column Nutrient Concentration

Water column total nitrogen and nitrate concentrations for Horsepen decreased exponentially downstream from the sewage treatment plant (R^2=0.813 and 0.772; both p<0.001, respectively) (Fig. 4), suggesting a sink for these nutrients (Fig. 2) via plant uptake, sediment adsorption or pore water retention. Ammonium concentrations also decreased exponentially (R^2=0.735; p<0.001) but the magnitude of this change was relatively small (Fig. 4). Mean total nitrogen, nitrate and ammonium concentrations decreased by 67, 75 and 16%, respectively, from station HB7 (~160 m) closest to the sewage outfall to station HB4 (~1300 m) and by 95, 99 and 96%, respectively, from station HB7 to HB1 (~2220 m) furthest from the sewage outfall. Water column nitrite and phosphate concentrations decreased linearly downstream from the nutrient source (R^2=0.650 and 0.793; both p<0.001) (Fig. 4), indicating that these constituents are diluted within the water column (Fig. 2). Mean nitrite concentrations decreased by 41% from station HB7 to HB3 (~1600 m) and by 99% from station HB7 to HB1. Mean phosphate concentrations decreased by 75% from station HB7 to HB3 and by 83% from station HB7 to HB1.

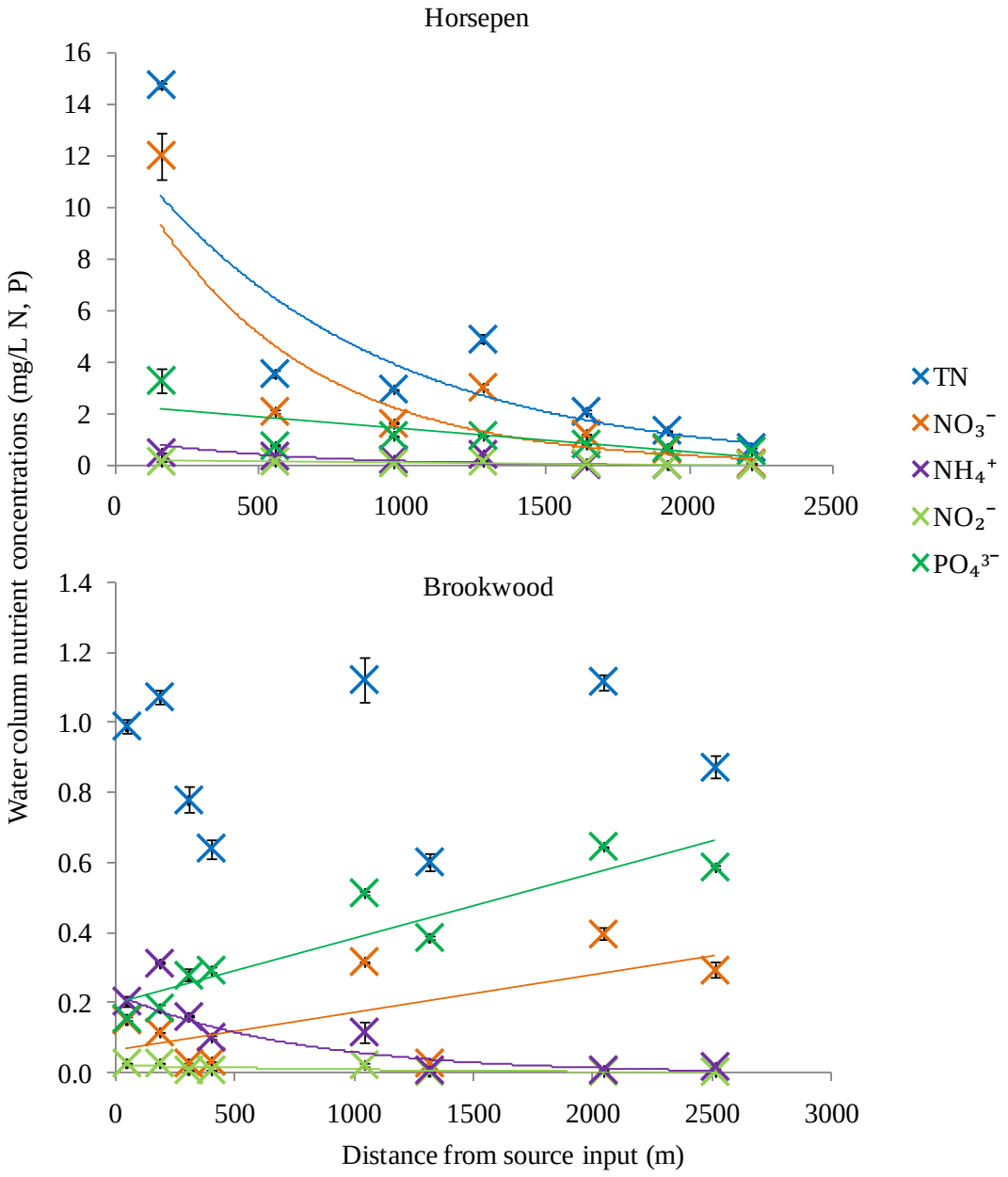

Fig. 4 Mean water column nutrient concentrations (mg/L N, P) measured from stations downstream from nutrient source input. Bars signify standard error (SE). *n*=3.

Water column total nitrogen concentrations for Brookwood were variable and did not change over distance (Fig. 4 and Table 1). Ammonium concentrations decreased exponentially (R^2=0.707; p<0.001) downstream from the detention basin (Fig. 4), suggesting this constituent is being removed from the system via plant uptake, sediment adsorption, or pore water retention. Nitrite concentrations also decreased (R^2=0.444; p<0.001) downstream from the detention basin but the magnitude of this change was very small (Fig. 4). Mean ammonium concentrations decreased by 97% from the detention basin to station BW3 (~1300 m) and by 93% from the detention basin to station BW1 (~2500 m) furthest from the detention basin. Mean nitrite concentrations decreased by 86% from the detention basin to station BW3 and by 75% from the detention basin to station BW1. Phosphate and nitrate concentrations increased linearly (R^2=0.844 and 0.417; both p<0.001) downstream from the detention basin (Fig. 4), suggesting an additional nutrient source downstream. Mean phosphate concentrations increased by 56% from the detention basin to station BW3 and by 71% from the detention basin to station BW1. Mean nitrate concentrations increased by 57% from the detention basin to BW3 and by 54% from the detention basin to station BW1.

Table 1 Least squares linear regression and exponential curve estimation between water column nutrient concentrations and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary		Water column nutrients	R^2	p
Horsepen	linear	TN (3)	0.601	<0.001
		NO_3^- (3)	0.549	<0.001
		NO_2^- (3)	0.650	<0.001
		NH_4^+ (3)	0.720	0.010
		PO_4^{3-} (3)	0.793	<0.001
	curve	TN (3)	0.813	<0.001
		NO_3^- (3)	0.772	<0.001
		NO_2^- (3)	0.585	<0.001
		NH_4^+ (3)	0.735	<0.001
		PO_4^{3-} (3)	0.597	<0.001
Brookwood	linear	TN (3)	0.386	0.001
		NO_3^- (3)	0.417	<0.001
		NO_2^- (3)	0.444	<0.001
		NH_4^+ (3)	0.623	<0.001
		PO_4^{3-} (3)	0.844	<0.001
	curve	TN (3)	-0.039	0.712
		NO_3^- (3)	0.210	0.014
		NO_2^- (3)	0.444	<0.001
		NH_4^+ (3)	0.707	<0.001
		PO_4^{3-} (3)	0.781	<0.001

3.1.2 Plant Tissue Nutrient Content

Aboveground/emergent plant tissue nutrient content for exotics, *A. philoxeroides* and *P. stratiotes* appeared constant and *P. stratiotes* did not change over distance for Horsepen (Fig. 5 and Table 2). This suggests that the plant tissues are saturated with nutrients and therefore not effective sinks to aid in water quality improvement downstream. Aboveground plant tissue total nitrogen for native *S. californicus* showed similar results (Fig. 5 and Table 2) but total phosphorus decreased (R^2=0.231; p=0.040), although the magnitude of this change was very small. Mean *S. californicus* aboveground plant tissue total phosphorus decreased by 17% from station HB6 (~550 m), the station closest to

sewage outfall this species was found, to station HB3 (~1600 m) and by 27% from station HB6 to station HB1. Aboveground/emergent plant tissue total nitrogen for Brookwood appeared constant and did not change over distance (Fig. 5 and Table 2), suggesting the plants along this tributary are also saturated and not effectively removing this nutrient for water quality improvement. *Schoenoplectus californicus* aboveground plant tissue total phosphorus for Brookwood decreased (R^2=0.301; p=0.025) but the magnitude of this change was very small (Fig. 5). *Alternanthera philoxeroides* aboveground plant tissue total phosphorus for Brookwood showed a small decrease but this occurred only across the four upstream stations; *A. philoxeroides* was not found at the four downstream stations (Fig. 5). This suggests this storage compartment for both species is not providing a sink for phosphorus. Mean *S. californicus* aboveground plant tissue total phosphorus decreased by 5% from station BW5 (~400 m), the station closest to source input this species was found, to station BW3 and by 39% from station BW5 to station BW1. Mean *A. philoxeroides* aboveground plant tissue total phosphorus decreased by 17% from inside the detention basin to station BW5, the furthest station from source input this species was found.

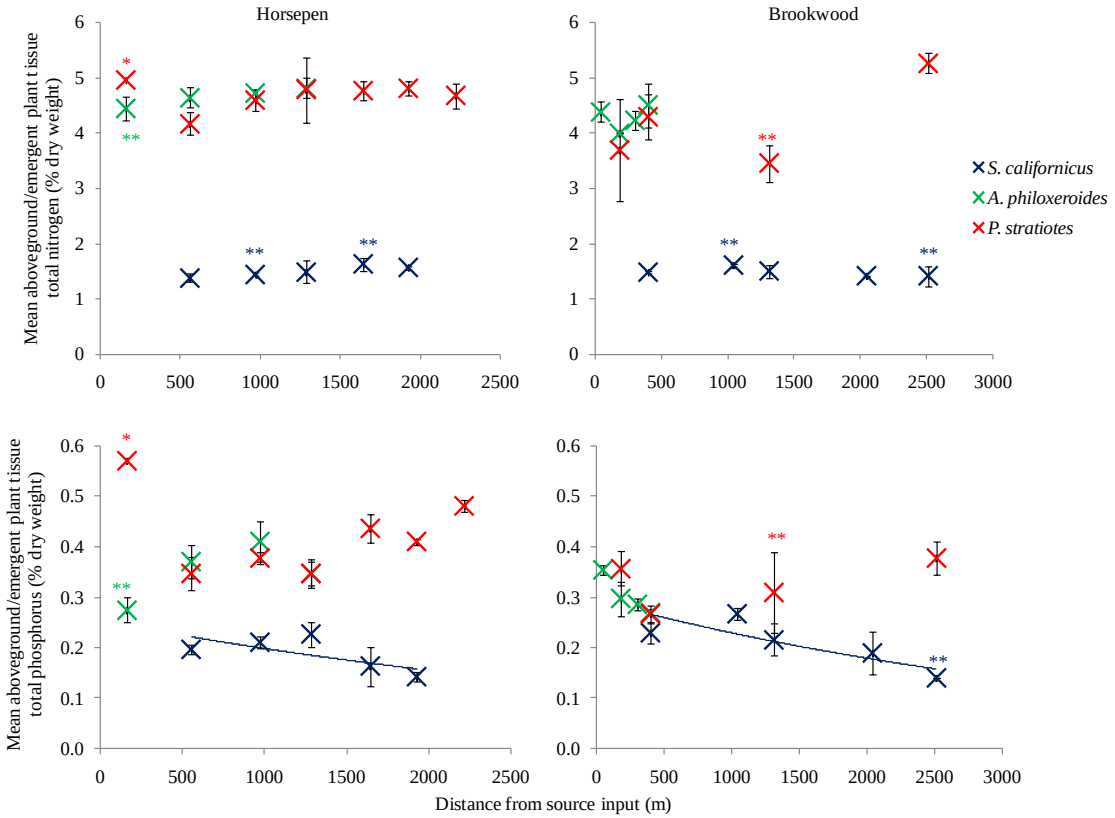

Fig. 5 Mean aboveground/emergent (stems/leaves) plant tissue total nitrogen and total phosphorus (% dry weight) of exotics *A. philoxeroides* and *P. stratiotes* and restored, native *S. californicus* measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 2 Least squares linear regression and exponential curve estimation between aboveground/emergent (stems/leaves) plant tissue total nitrogen (TN) and total phosphorus (TP) of exotic *P. stratiotes* and restored, native *S. californicus* and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			Nutrients	R^2	p
Horsepen	*S. californicus*	aboveground	linear	TN (13)	0.211	0.065
				TP (15)	0.177	0.066
			curve	TN	0.219	0.061
				TP	0.231	0.040
	P. stratiotes	emergent	linear	TN (19)	-0.006	0.359
				TP (19)	0.020	0.259
			curve	TN	-0.003	0.343
				TP	0.050	0.181
Brookwood	*S. californicus*	aboveground	linear	TN (13)	0.005	0.325
				TP (14)	0.282	0.029
			curve	TN	0.020	0.289
				TP	0.301	0.025
	P. stratiotes	emergent	linear	TN (11)	0.163	0.120
				TP (11)	0.027	0.288
			curve	TN	0.112	0.166
				TP	0.018	0.305

Native and exotic aboveground/emergent plant tissue nutrient content appeared to be very different for both tributaries (Fig. 5). *Schoenoplectus californicus* mean aboveground plant tissue total nitrogen and phosphorus for Horsepen were lower compared to exotics. Similar results were measured for Brookwood, suggesting exotics have a higher nutrient retention capacity than the native species.

Schoenoplectus californicus belowground plant tissue nutrient content for Horsepen remained fairly constant and did not change over distance (Fig. 6 and Table 3). This suggests this storage compartment is saturated and not contributing to overall water quality improvement. Similar results were observed for *Pistia stratiotes* submerged

plant tissue total nitrogen (Fig. 6 and Table 3). *Pistia stratiotes* submerged plant tissue total phosphorus linearly increased (R^2=0.447; p=0.003) with increasing distance but this change was small in magnitude (Fig. 6). Mean submerged plant tissue total phosphorus increased by 6% from station HB7 to station HB3 (~1600 m) and by 18% from station HB7 to station HB1. Belowground plant tissue total nitrogen content for *A. philoxeroides* did not vary with distance from the nutrient source in Horsepen, although this species only occurred in the two stations furthest upstream (<800 m) (Fig. 6). Belowground plant tissue total phosphorus content for *A. philoxeroides* was measured at only one station (closest to the sewage outfall) for Horsepen and it was similar *S. californicus* and lower than *P. stratiotes* (Fig. 6). *Alternanthera philoxeroides* belowground plant tissue nutrient content did not vary with distance from the nutrient source in Brookwood, although this species occurred only in the four stations furthest upstream (<400 m) (Fig. 6). *Schoenoplectus californicus* belowground plant tissue total nitrogen for Brookwood remained constant and did not change over distance (Fig. 6 and Table 3). This suggests this storage compartment is saturated and not an effective nitrogen sink. *Schoenoplectus californicus* belowground plant tissue total phosphorus decreased exponentially (R^2=0.734; p<0.001) with increased distance for Brookwood (Fig. 6), suggesting this species' storage compartment is not a sink for this constituent.

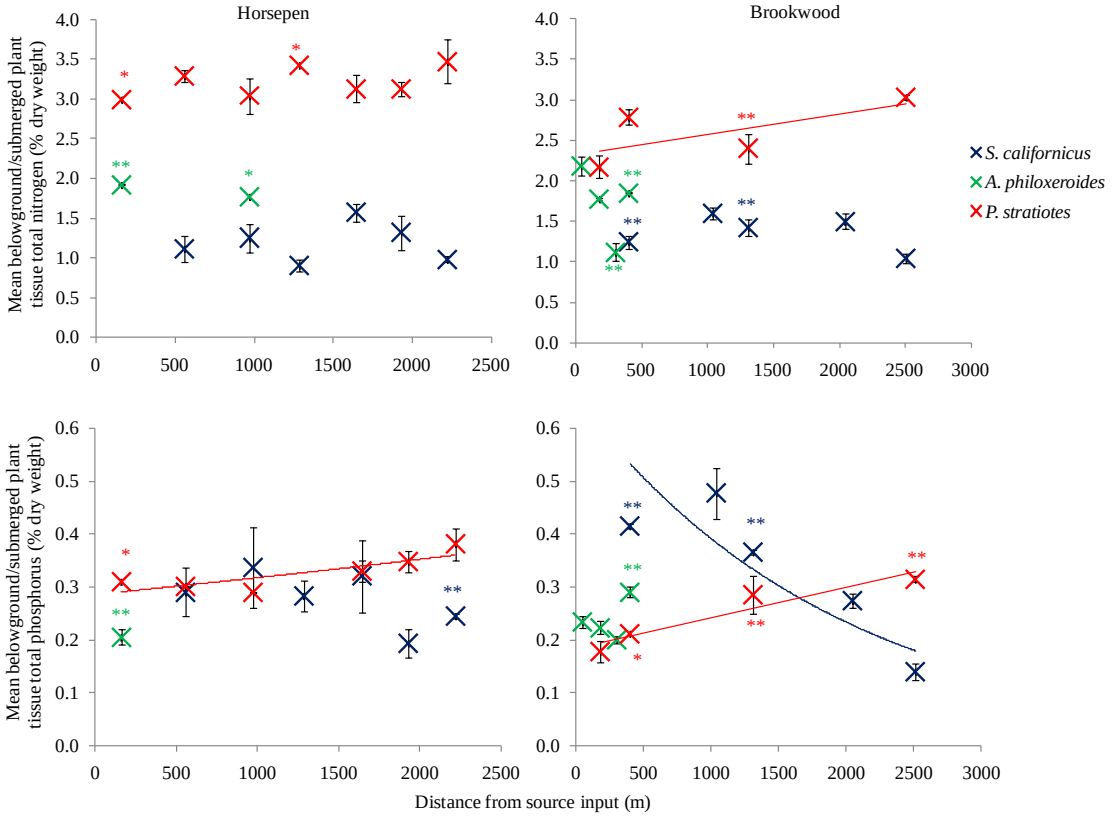

Fig. 6 Mean belowground/submerged (roots) plant tissue total nitrogen and total phosphorus (% dry weight) of exotics *A. philoxeroides* and *P. stratiotes* and restored, native *S. californicus* measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 3 Least squares linear regression and exponential curve estimation between belowground/submerged (roots) plant tissue total nitrogen (TN) and total phosphorus (TP) of exotic *P. stratiotes* and restored, native *S. californicus* and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			Nutrients	R^2	p
Horsepen	*S. californicus*	belowground	linear	TN (17)	-0.050	0.636
				TP (17)	0.052	0.191
			curve	TN	-0.052	0.658
				TP	0.062	0.172
	P. stratiotes	submerged	linear	TN (17)	-0.015	0.395
				TP (16)	0.428	0.004
			curve	TN	-0.019	0.414
				TP	0.447	0.003
Brookwood	*S. californicus*	belowground	linear	TN (13)	0.059	0.213
				TP (13)	0.729	<0.001
			curve	TN	0.082	0.177
				TP	0.734	<0.001
	P. stratiotes	submerged	linear	TN (11)	0.348	0.033
				TP (8)	0.738	0.004
			curve	TN	0.321	0.040
				TP	0.679	0.007

Schoenoplectus californicus mean belowground plant tissue total phosphorus decreased by 12% from station BW5 (~400 m) to station BW3 and by 66% from station BW5 to station BW1. *Pistia stratiotes* submerged plant tissue total nitrogen and phosphorus increased linearly (R^2=0.348 and 0.738; p=0.033 and 0.004, respectively) with distance (Fig. 6), although this change was relatively small in magnitude. Mean submerged plant tissue total nitrogen increased by 9% from station BW7 to station BW3 and by 28% from station BW7 to station BW1. Mean submerged plant tissue total phosphorus increased by 38% from station BW7 to station BW3 and by 44% from station BW7 to station BW1.

Native and exotic belowground/submerged plant tissue had very different total nitrogen content, but total phosphorous was similar for both tributaries (Fig. 6). *Schoenoplectus californicus* mean belowground plant tissue total nitrogen for Horsepen was lower compared to exotics *P. stratiotes* and *A. philoxeroides*. *Schoenoplectus californicus* belowground plant tissue total phosphorus for Horsepen was similar to exotics *P. stratiotes* and *A. philoxeroides*. Brookwood showed similar results between native *S. californicus* and exotics, *A. philoxeroides* and *P. stratiotes*. This suggests the belowground/submerged storage compartment for exotics is more effective at removing nitrogen from the water column. Conversely, both the native and exotics appear to have similar phosphorus storage capacities in the belowground/submerged plant tissue.

Aboveground/emergent plant tissue C/N ratio for exotics remained uniform and *P. stratiotes* did not change with increasing distance for both tributaries (Fig. 7 and Table 4). The nutrient limitation thresholds in Fig. 7 are based on the median C/N/P ratios for aquatic angiosperms (500:24:1) reported by Duarte (1992) and suggest that the native bulrush is nitrogen limited, but that the exotic species are not. Native *S. californicus* aboveground plant tissue C/N ratio was stable and did not change over distance for Horsepen, but relatively high mean C/N ratios across all stations (28.4±1.0 SE) suggest that nitrogen may be limiting (Fig. 7 and Table 4). Similar results were observed for Brookwood. Aboveground plant tissue C/N ratio for *S. californicus* did not

change with increasing distance for Brookwood, but relatively high mean C/N ratios across all stations (28.6±1.0 SE) suggest that this nutrient is limiting in this storage compartment (Fig. 7 and Table 4). Aboveground/emergent plant tissue C/P ratios for all three species along Horsepen appeared uniform with increasing distance and were well below the phosphorus limitation threshold, suggesting all three species are not phosphorus limited in this storage compartment (Fig. 7). *Schoenoplectus californicus* aboveground plant tissue C/P ratio for Brookwood increased exponentially (R^2=0.359; p=0.014) suggesting a decrease in phosphorus supply, although this species was well below the phosphorus limitation threshold (Fig. 7). *Pistia stratiotes* emergent plant tissue appeared to not be phosphorus limited in this storage compartment (Fig. 7). *Alternanthera philoxeroides* appeared to increase in aboveground plant tissue C/P ratio with increasing distance for Brookwood although this occurred across only four stations (<400 m) and was still well below the phosphorus limitation threshold (Fig. 7).

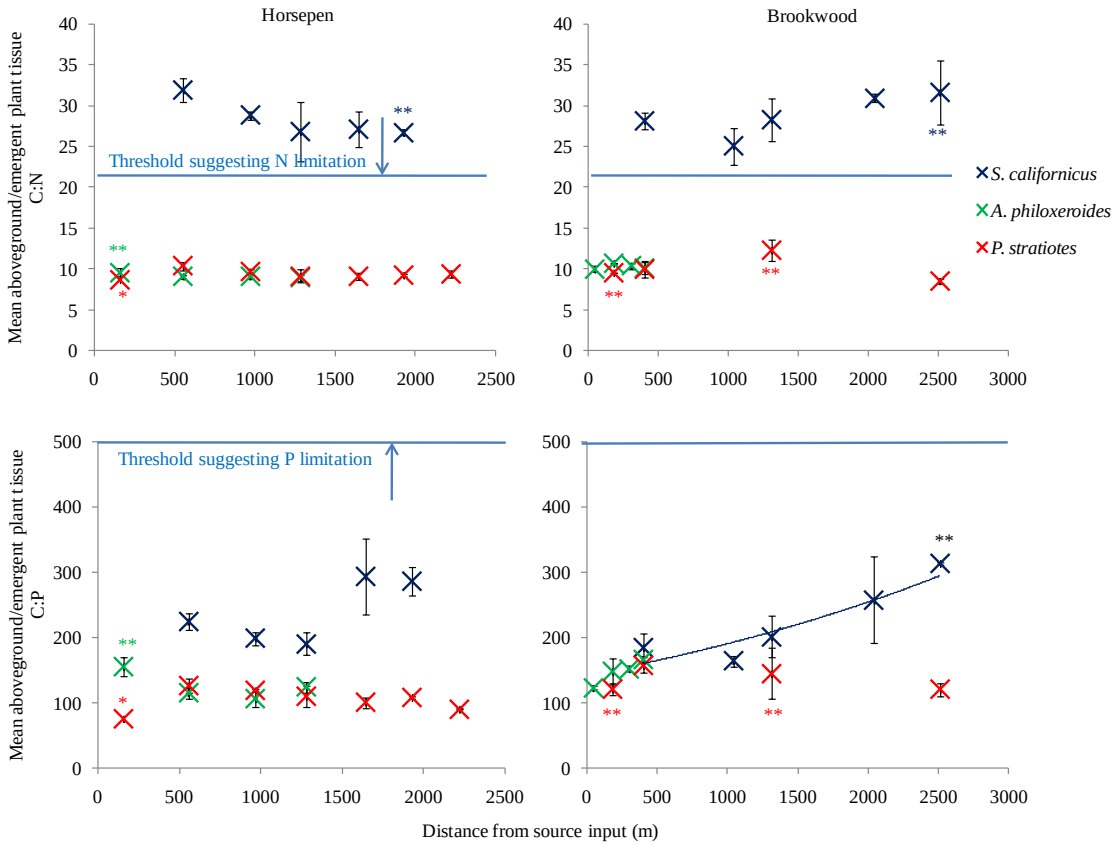

Fig. 7 Mean aboveground/emergent (stems/leaves) plant tissue C/N and C/P ratios of exotics *A. philoxeroides* and *P. stratiotes* and restored, native *S. californicus* measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 4 Least squares linear regression and exponential curve estimation between aboveground/emergent (stems/leaves) plant tissue C/N and C/P ratios of exotic *P. stratiotes* and restored, native *S. californicus* and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			Ratio	R^2	p
Horsepen	*S. californicus*	aboveground	linear	C:N (14)	0.185	0.070
				C:P (15)	0.194	0.057
			curve	C:N	0.163	0.085
				C:P	0.164	0.075
	P. stratiotes	emergent	linear	C:N (19)	0.004	0.316
				C:P (19)	0.046	0.188
			curve	C:N	-0.002	0.341
				C:P	0.026	0.241
Brookwood	*S. californicus*	aboveground	linear	C:N (14)	0.142	0.101
				C:P (14)	0.352	0.015
			curve	C:N	0.113	0.129
				C:P	0.359	0.014
	P. stratiotes	emergent	linear	C:N (10)	-0.047	0.462
				C:P (11)	-0.046	0.474
			curve	C:N	-0.023	0.398
				C:P	-0.039	0.450

Belowground/submerged plant tissue C/N ratios for exotics were uniform with increasing distance for Horsepen although *A. philoxeroides* occurred across only two stations (<800 m) (Fig. 8). Across all stations, *P. stratiotes* and *A. philoxeroides* had low C/N ratios (11.4±0.2 and 20.8±1.3 SE, respectively), suggesting these exotics are not nitrogen limited in this storage compartment. *Pistia stratiotes* belowground plant tissue C/N ratio did not change with increasing distance for Horsepen (Table 5). *Schoenoplectus californicus* belowground plant tissue C/N ratio was uniform and did not change with increasing distance for Horsepen (Fig. 8 and Table 5). The restored native species had relatively high mean belowground plant tissue C/N ratios across all stations

(34.4±2.4 SE), suggesting nitrogen limitation within this storage compartment. *Pistia stratiotes* submerged plant tissue C/P ratio decreased (R^2=0.614; p<0.001) with increasing distance for Horsepen but the magnitude of this change was small (Fig. 8). *Alternanthera philoxeroides* belowground tissue C/P ratio was measured at only one station (closest to the sewage outfall) for Horsepen and it was higher compared to the other species (Fig. 8). *Schoenoplectus californicus* belowground plant tissue C/P ratio was stable with increasing distance for Horsepen and relatively low across all stations (143.2±10.4 SE). *Schoenoplectus californicus* and *P. stratiotes* belowground/submerged plant tissue C/N ratios for Brookwood did not change with increasing distance (Fig. 8 and Table 5). *Schoenoplectus californicus* had relatively high mean belowground plant tissue C/N ratios across all stations (27.2±1.6 SE) suggesting nitrogen limitation for Brookwood. *Pistia stratiotes* and *A. philoxeroides* had low mean belowground plant tissue C/N ratios across all stations (13.1±0.9 and 20.5±1.5 SE, respectively) suggesting both exotics are not nitrogen limited in this storage compartment. *Schoenoplectus californi*cus belowground plant tissue C/P ratio increased linearly (R^2=0.664; p=0.001) with increasing distance for Brookwood, but relatively low mean C/P ratios across all stations (112.5±13.4 SE) suggest this storage compartment is not phosphorus limited (Fig. 8). *Pistia stratiotes* submerged plant tissue C/P ratio decreased exponentially (R^2=0.478; p=0.035) with increasing distance, suggesting this storage compartment is not phosphorus limited (Fig. 8). Belowground plant tissue C/P ratios for *A. philoxeroides* appeared to decrease with increasing distance, suggesting this storage

compartment is not phosphorus limited, but this species occurred only at the four stations furthest upstream (Fig. 8).

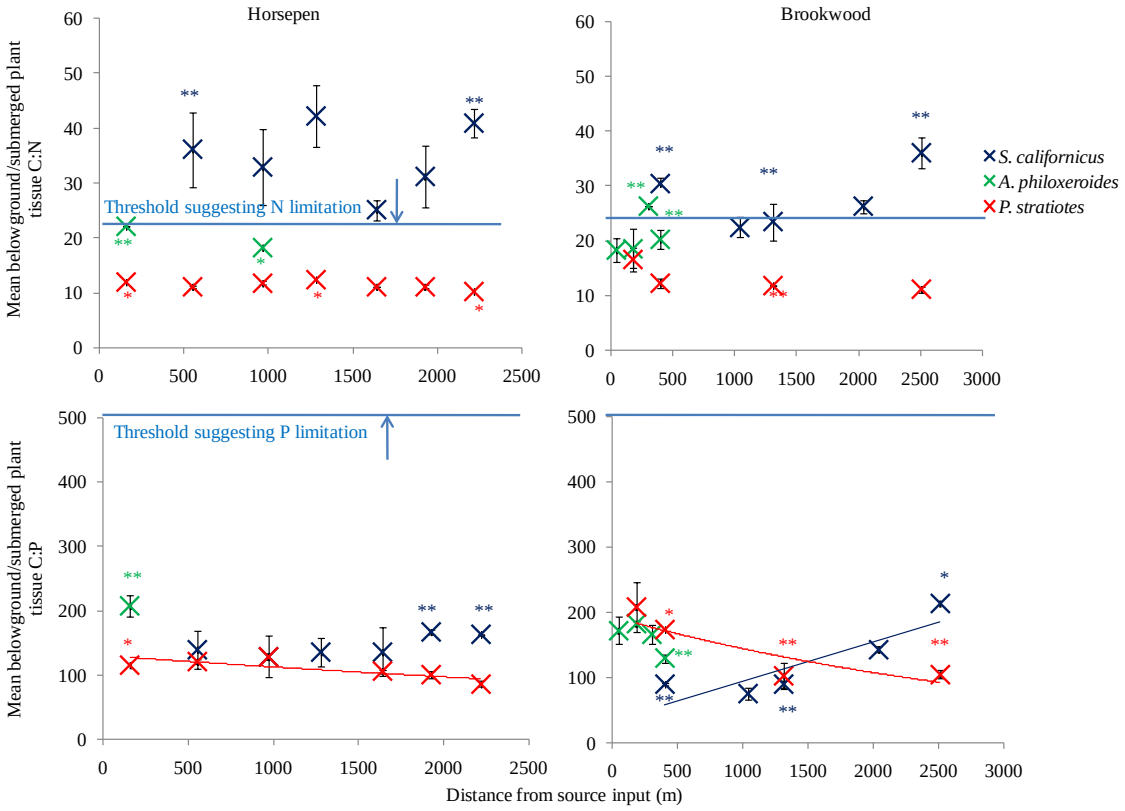

Fig. 8 Mean belowground/submerged (roots) plant tissue C/N and C/P ratios of exotics *A. philoxeroides* and *P. stratiotes* and restored, native *S. californicus* measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 5 Least squares linear regression and exponential curve estimation between belowground/submerged (roots) plant tissue C/N and C/P ratios of exotic *P. stratiotes* and restored, native *S. californicus* and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			Ratio	R^2	p
Horsepen	*S. californicus*	belowground	linear	C:N (17)	-0.061	0.785
				C:P (16)	-0.012	0.381
			curve	C:N	-0.063	0.825
				C:P	0.000	0.337
	P. stratiotes	submerged	linear	C:N (14)	0.099	0.145
				C:P (16)	0.606	<0.001
			curve	C:N	0.104	0.139
				C:P	0.614	<0.001
Brookwood	*S. californicus*	belowground	linear	C:N (12)	0.058	0.225
				C:P (11)	0.664	0.001
			curve	C:N	0.038	0.259
				C:P	0.622	0.002
	P. stratiotes	submerged	linear	C:N (11)	0.246	0.069
				C:P (8)	0.435	0.045
			curve	C:N	0.276	0.056
				C:P	0.478	0.035

3.1.3 Plant Biomass

Plant total biomass was variable for *S. californicus* and did not significantly change with increasing distance from source input for both tributaries (Fig. 9 and Table 6). *Alternanthera philoxeroides* biomass decreased with increasing distance from source input for both tributaries although this species only occurred at two stations in Horsepen (<800 m) and four stations in Brookwood (<400 m) (Fig. 9). This suggests that this species' storage compartment is not storing excess nutrient uptake in the form of biomass. Biomass for *P. stratiotes* was very low for both tributaries and did not change with increasing distance (Fig. 9 and Table 6). This indicates that this species does not

provide a sufficient storage compartment for excess nutrients and is therefore not substantially contributing to water quality improvement.

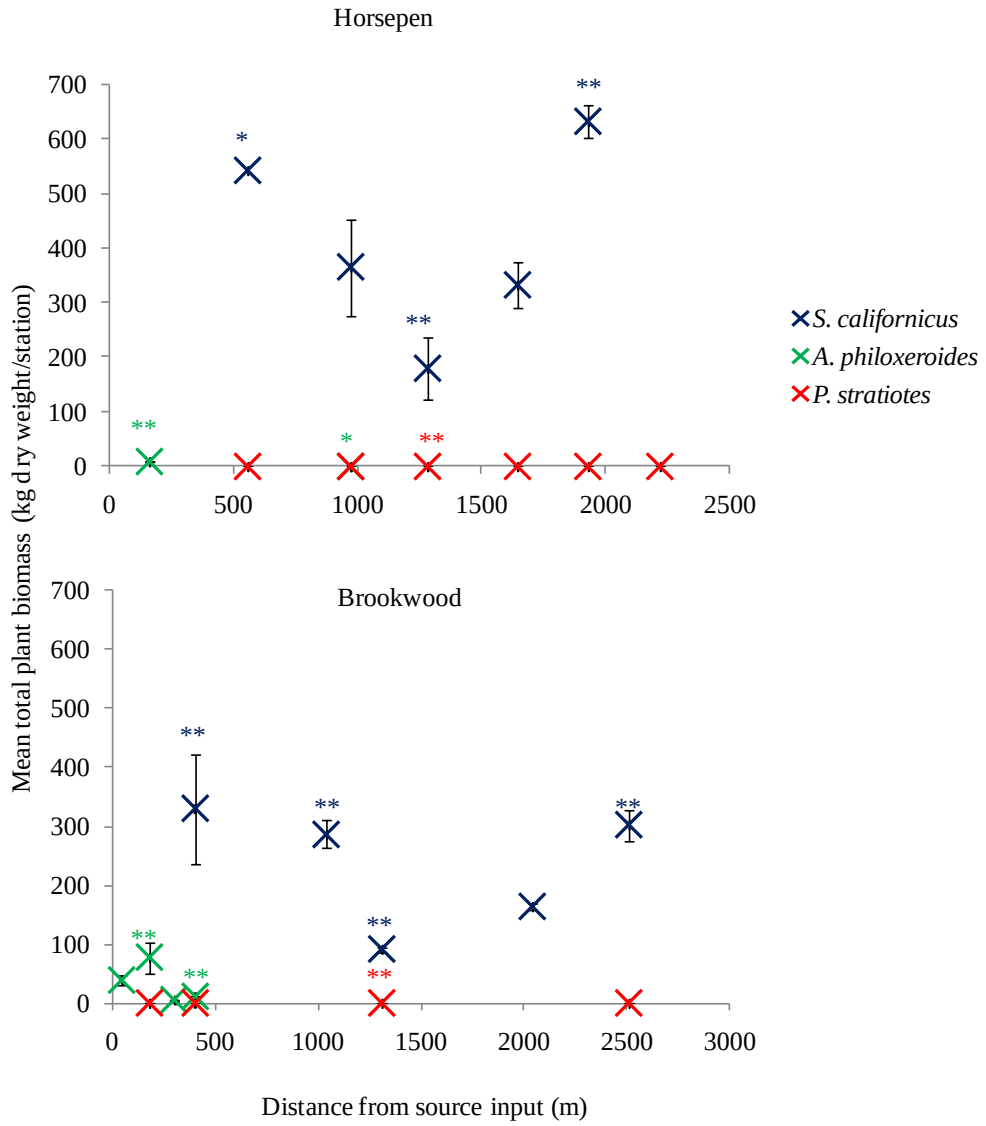

Fig. 9 Mean plant total biomass (kg dry weight/station) of exotics *A. philoxeroides* and *P. stratiotes* and restored, native *S. californicus* measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 6 Least squares linear regression and exponential curve estimation between total plant biomass (kg dry weight/station) of exotic *P. stratiotes* and restored, native *S. californicus* and distance from nutrient source input. Based on confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			R^2	p
Horsepen	*S. californicus*	total plant biomass (11)	linear	-0.062	0.536
			curve	-0.079	0.615
	P. stratiotes	total plant biomass (18)	linear	0.058	0.173
			curve	0.113	0.095
Brookwood	*S. californicus*	total plant biomass (11)	linear	-0.053	0.500
			curve	-0.089	0.680
	P. stratiotes	total plant biomass (11)	linear	0.178	0.109
			curve	0.195	0.098

3.1.4 Stem Density

Stem density for *S. californicus* and *A. philoxeroides* did not change with increasing distance in Horsepen (Fig. 10 and Table 7). Stem density for *P. stratiotes* increased (R^2=0.575; $p<0.001$) with increasing distance although the magnitude of change was relatively small (Fig. 10). This suggests this species is not providing a sufficient storage compartment (in terms of biomass) for nutrients to contribute to overall water quality improvement. *Schoenoplectus californicus* and *P. stratiotes* stem density did not change with increasing distance for Brookwood (Fig. 10 and Table 7). Stem density for *A. philoxeroides* decreased with increasing distance from source input although this trend was only observed across 4 stations (<400 m) (Fig. 10). This suggests a lack of plant biomass for effective nutrient removal to aid in water quality improvement.

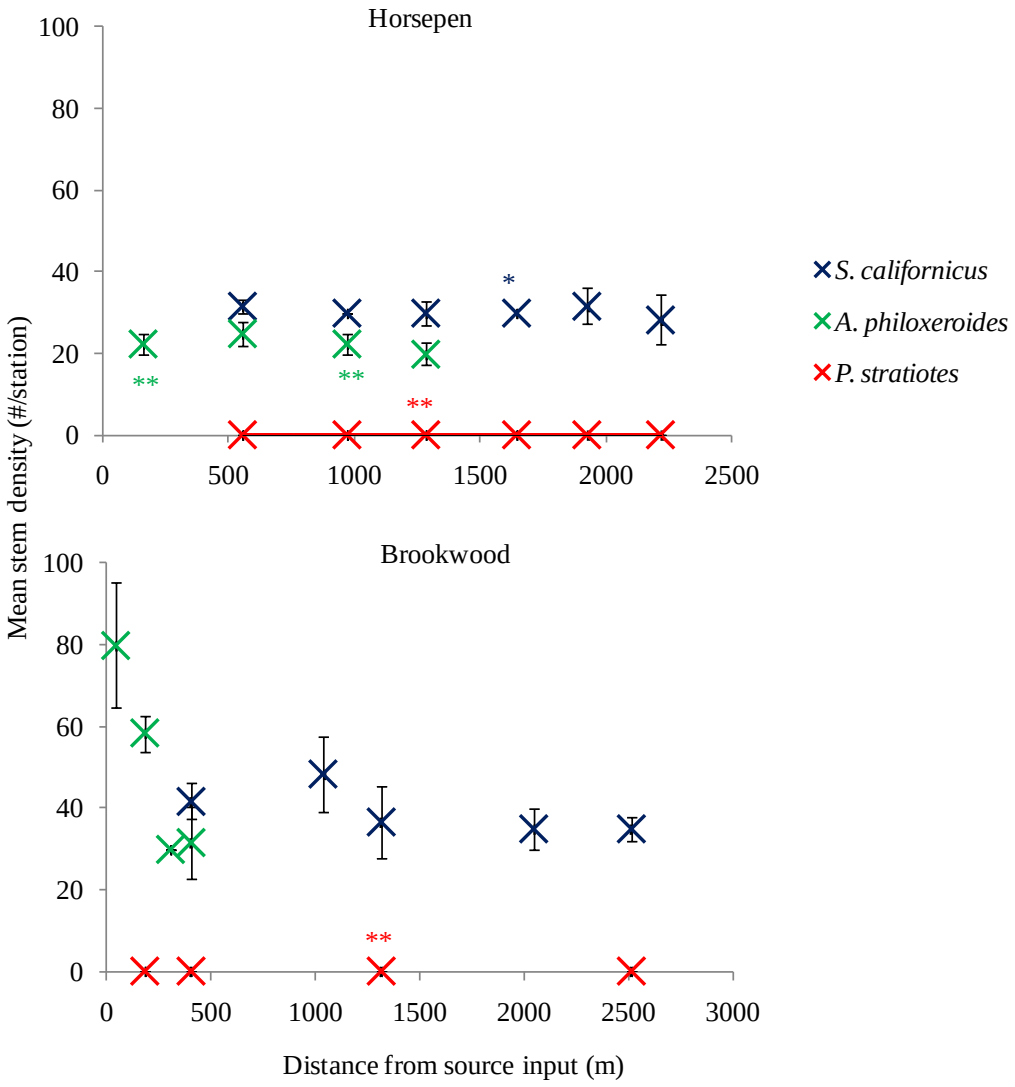

Fig. 10 Mean stem density (#/station) of exotics *A. philoxeroides* and *P. stratiotes* and restored, native *S. californicus* measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 7 Least squares linear regression and exponential curve estimation between stem density of exotic *P. stratiotes* and restored, native *S. californicus* and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			R^2	p
Horsepen	*S. californicus*	stem density (16)	linear	-0.057	0.666
			curve	-0.031	0.473
	P. stratiotes	stem density (17)	linear	0.477	0.001
			curve	0.575	<0.001
Brookwood	*S. californicus*	stem density (15)	linear	0.039	0.233
			curve	0.016	0.289
	P. stratiotes	stem density (11)	linear	0.012	0.317
			curve	-0.092	0.699

3.1.5 Pore Water Nutrient Concentration

Regression analysis showed pore water nitrite for Horsepen decreased exponentially (R^2=0.748; p=0.007) with increased distance (Fig. 11), suggesting pore water is not providing a sink for this constituent. Mean nitrite concentrations decreased by 94% from station HB7 to station HB4 and by 97% from station HB7 to station HB1. Phosphate concentrations increased exponentially (R^2=0.527; p=0.039) with increased distance although the absolute change was quite small (Fig. 11). Mean phosphate concentrations increased by 93% from station HB7 to station HB4 and by 98% from station HB7 to station HB1. Pore water total nitrogen, nitrate and ammonium concentrations for Horsepen were variable and did not significantly change with increased distance, suggesting pore water may not be a sufficient sink for these nutrients (Fig. 11 and Table 8).

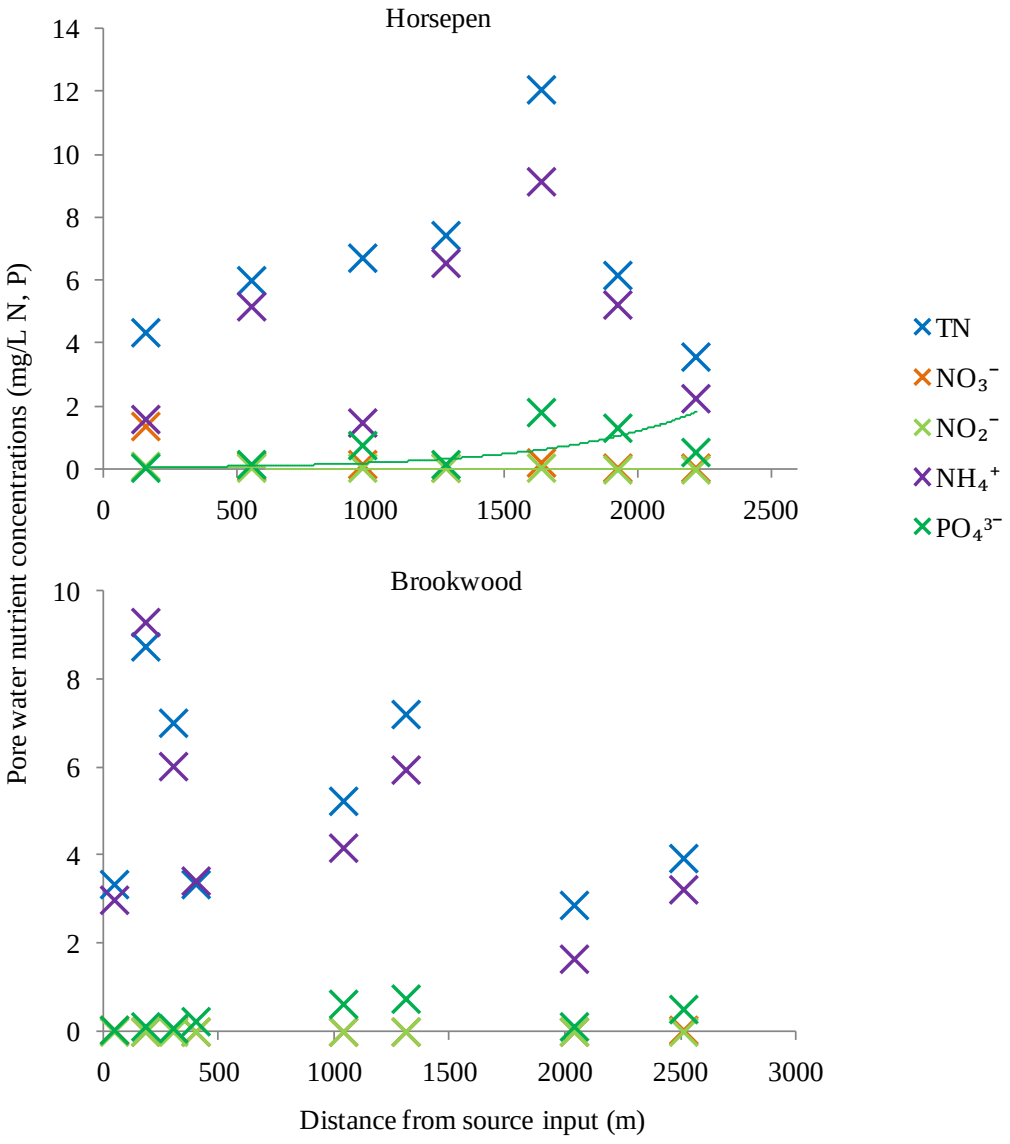

Fig. 11 Pore water nutrient concentrations (mg/L N, P) measured from stations downstream from nutrient source input. *n*=1.

Table 8 Least squares linear regression and exponential curve estimation between pore water nutrient concentrations and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary		Pore water nutrients	R^2	p
Horsepen	linear	TN (1)	-0.174	0.754
		NO_3^- (1)	0.357	0.092
		NO_2^- (1)	0.427	0.066
		NH_4^+ (1)	-0.096	0.522
		PO_4^{3-} (1)	0.237	0.151
	curve	TN (1)	-0.194	0.883
		NO_3^- (1)	0.409	0.072
		NO_2^- (1)	0.748	0.007
		NH_4^+ (1)	-0.066	0.464
		PO_4^{3-} (1)	0.527	0.039
Brookwood	linear	TN (1)	-0.021	0.391
		NO_3^- (1)	-0.118	0.628
		NO_2^- (1)	0.120	0.211
		NH_4^+ (1)	0.096	0.235
		PO_4^{3-} (1)	0.134	0.199
	curve	TN (1)	-0.039	0.424
		NO_3^- (1)	-0.117	0.624
		NO_2^- (1)	0.100	0.230
		NH_4^+ (1)	0.131	0.201
		PO_4^{3-} (1)	0.219	0.136

In Brookwood, all pore water nutrient concentrations did not change with increased distance (Fig. 11 and Table 8), suggesting that pore water is not providing a sink for water column nutrients for this tributary. Total nitrogen and ammonium concentrations were variable for Brookwood. Nitrate, nitrite and phosphate concentrations were very low throughout the tributary.

3.1.6 Sediment Nutrient and Organic Content

Sediment total nitrogen and phosphorus contents were low for Horsepen (0.076±0.008% N and 0.02±0.001 SE % P dry weight) and Brookwood (0.177±0.018% N and 0.03±0.002 SE % P dry weight). Sediment total nitrogen increased (R^2=0.531; $p<0.001$)

for Horsepen but the magnitude of change was small, suggesting that this wetland storage compartment is not an effective sink to contribute to overall water quality improvement downstream (Fig. 12). Mean sediment total nitrogen increased by 26% from HB7 to station HB4 and by 61% from station HB7 to station HB1. Mean total phosphorus remained fairly constant (Fig. 12) for Horsepen and did not change with increased distance (Table 9). Brookwood sediment total nitrogen decreased exponentially (R^2=0.236; p=0.009), suggesting that sediment is not a sink for nitrogen along this tributary (Fig. 12). Mean sediment total nitrogen decreased by 11% from inside the detention basin to station BW3 and by 45% from the detention basin to station BW1. Sediment total phosphorus linearly decreased (R^2=0.199; p=0.017) (Fig. 12), suggesting that this storage compartment is not an important sink for this constituent. Mean sediment total phosphorus decreased by 19% from the detention basin to station BW3 and by 29% from the detention basin to station BW1.

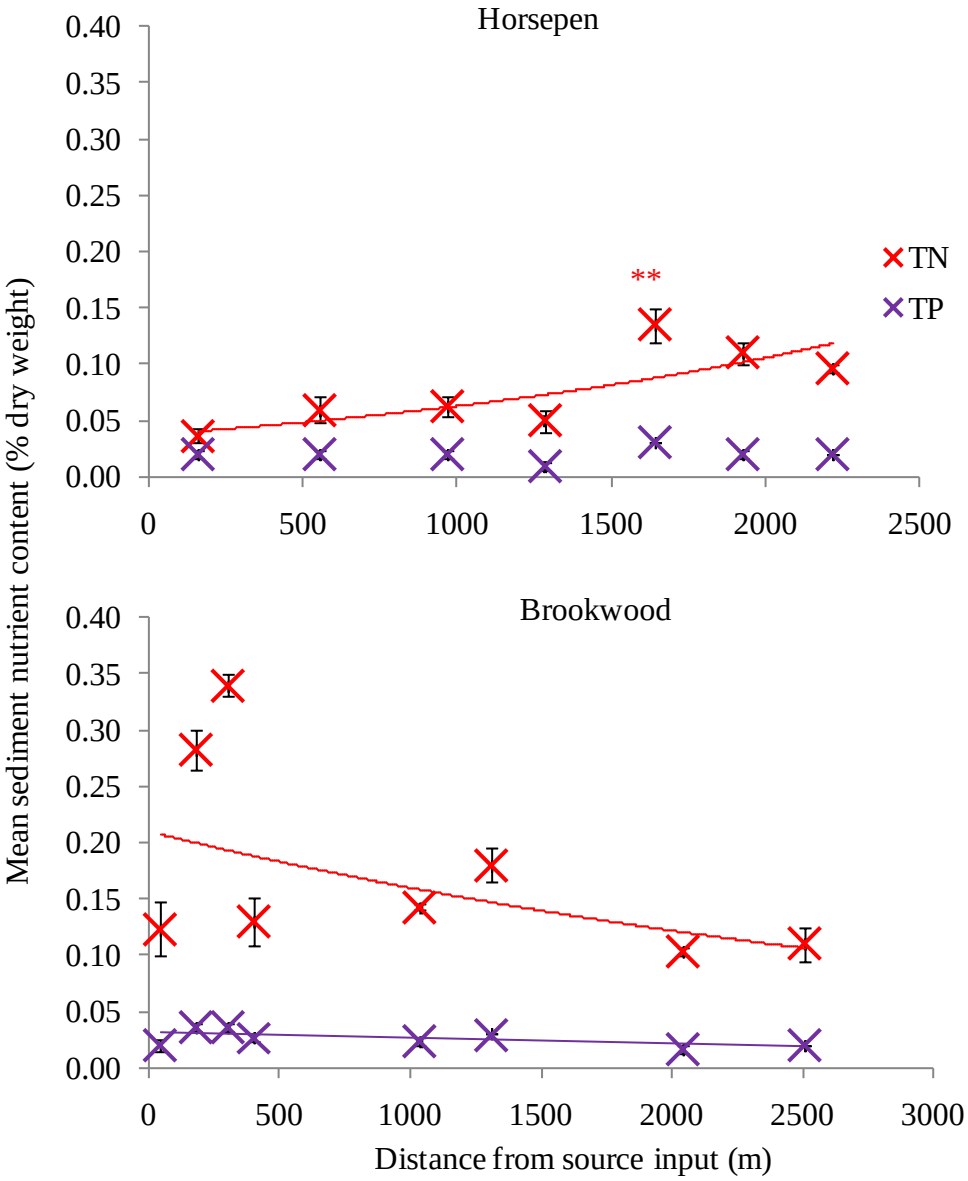

Fig. 12 Mean sediment total nitrogen and total phosphorus content (% dry weight) measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 9 Least squares linear regression and exponential curve estimation between sediment total nitrogen and total phosphorus distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary		Sediment	R^2	p
Horsepen	linear	TN (20)	0.510	<0.001
		TP (21)	-0.052	0.891
	curve	TN	0.531	<0.001
		TP	-0.052	0.899
Brookwood	linear	TN (24)	0.230	0.010
		TP (24)	0.199	0.017
	curve	TN	0.236	0.009
		TP	0.132	0.045

Regression analysis showed sediment organic content increased (R^2=0.313; p=0.005) for Horsepen but the change was small in magnitude, and decreased exponentially (R^2=0.434; p<0.001) for Brookwood (Fig. 13 and Table 10). This suggests the potential for increased sediment nutrient adsorption for Horsepen and decreased nutrient adsorption for Brookwood. Sediment consisted mostly of silty clay (45.273±2.628 SE %) for Horsepen and Brookwood (58.114±2.782 SE %) compared to sand (27.503±1.455 and 34.443±1.262 SE %, respectively). Regression analysis showed no significant change in sediment grain size with increasing distance from source input (Table 10).

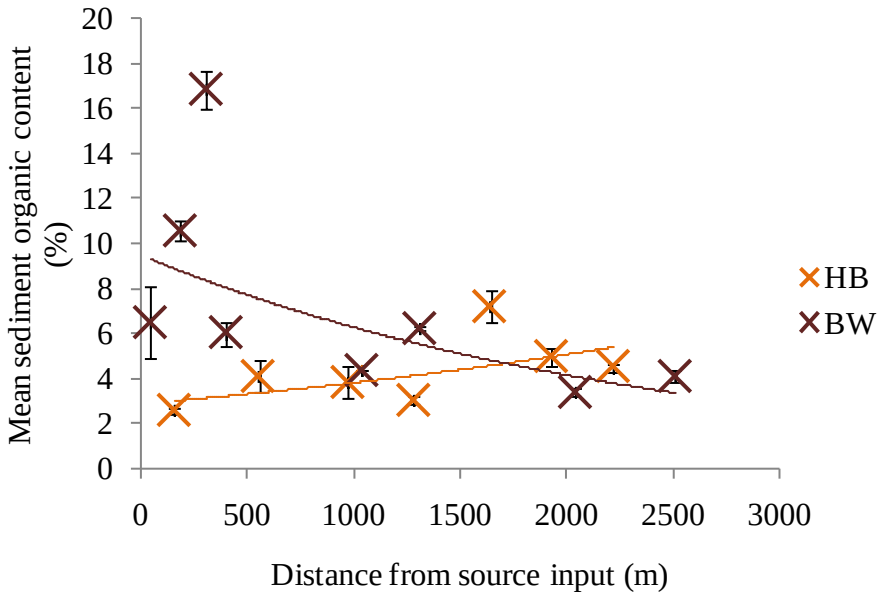

Fig. 13 Mean sediment organic content (%) measured from stations downstream from nutrient source input. Bars signify standard error (SE). *n*=3.

Table 10 Least squares linear regression and exponential curve estimation between sediment organic content (OC) and grain size and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary			R^2	p
Horsepen	linear	OC (21)	0.226	0.017
		silt+clay (18)	-0.043	0.594
		sand (18)	-0.043	0.593
	curve	OC	0.313	0.005
		silt+clay	-0.041	0.570
		sand	-0.046	0.625
Brookwood	linear	OC (24)	0.310	0.003
		silt+clay (21)	-0.051	0.883
		sand (21)	-0.051	0.883
	curve	OC	0.434	<0.001
		silt+clay	-0.052	0.939
		sand	-0.050	0.823

3.2 Armand Bayou Bioassay (ABBA)

3.2.1 *E. crassipes* Plant Tissue Nutrient Content

Emergent and submerged plant tissue total nitrogen for Horsepen was fairly constant and did not change with increased distance from source input (Fig. 14 and Table 11). This suggests *E. crassipes* has become saturated and therefore not providing an effective sink for nitrogen to contribute to overall water quality improvement for Horsepen. Submerged plant tissue total phosphorus did not change with increased distance (Fig. 14 and Table 11) suggesting saturation in this nutrient storage compartment. Emergent plant tissue total phosphorus for Horsepen decreased linearly (R^2=0.567; p=0.001) suggesting this storage compartment is not a sink for this constituent (Fig. 14). Emergent plant tissue nutrient content for Brookwood did not change with increased distance (Fig. 14 and Table 11) suggesting nutrient saturation in this storage compartment. Submerged plant tissue total nitrogen and phosphorus increased linearly (R^2=0.356 and 0.608; p=0.009 and <0.001, respectively) suggesting nutrient uptake in this nutrient storage compartment. Loss of all samples at station BW7 after deployment resulted in elimination of this station from the analysis.

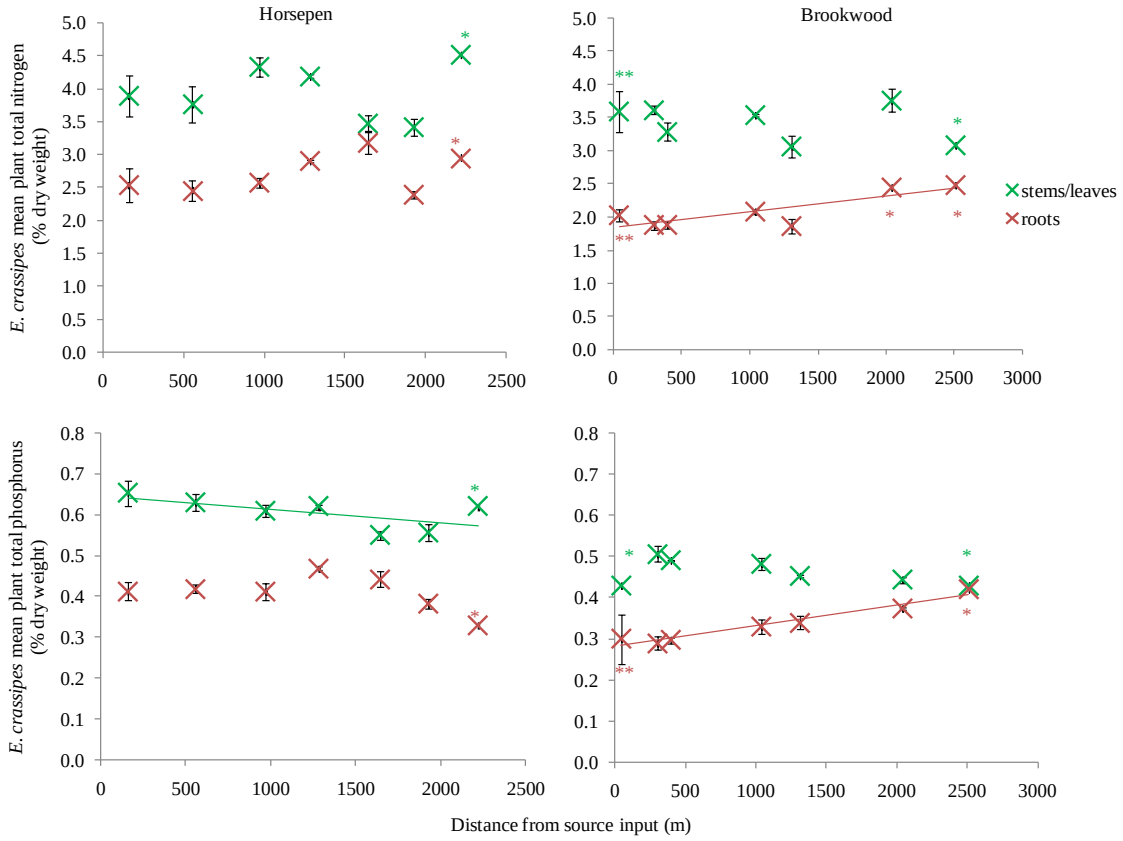

Fig. 14 *E. crassipes* mean emergent (stems/leaves) and submerged (roots) plant tissue total nitrogen and total phosphorus (% dry weight) measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 11 Least squares linear regression and exponential curve estimation between *E. crassipes* emergent (stems/leaves) and submerged (roots) plant tissue total nitrogen and total phosphorus (% dry weight) and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			Nutrients	R^2	p
Horsepen	*E. crassipes*	stems/leaves	linear	TN (19)	-0.028	0.485
				TP (19)	0.394	0.002
			curve	TN	-0.026	0.468
				TP	0.390	0.003
		roots	linear	TN (19)	0.050	0.180
				TP (19)	-0.012	0.387
			curve	TN	0.051	0.179
				TP	0.000	0.333
Brookwood	*E. crassipes*	stems/leaves	linear	TN (18)	-0.058	0.806
				TP (17)	0.143	0.075
			curve	TN	-0.056	0.758
				TP	0.138	0.079
		roots	linear	TN (16)	0.356	0.009
				TP (18)	0.608	<0.001
			curve	TN	0.311	0.014
				TP	0.580	<0.001

The nutrient limitation threshold depicted in Fig. 15 suggests *E. crassipes* is not limited in nitrogen or phosphorus content. Emergent and submerged plant tissue C/N and C/P ratios for Horsepen did not change over increasing distance, suggesting this species is not nutrient limited along this tributary (Fig. 15 and Table 12). Emergent plant tissue C/N and C/P ratios appeared constant and did not show a trend with increasing distance for Brookwood (Fig. 15 and Table 12) suggesting this species is not nitrogen or phosphorus limited in this storage compartment for this tributary. Submerged plant tissue C/N and C/P ratios decreased (R^2=0.470 and 0.270; p=0.001 and

0.019, respectively) although the absolute change was small. This suggests this compartment is not nutrient limited (Fig. 15).

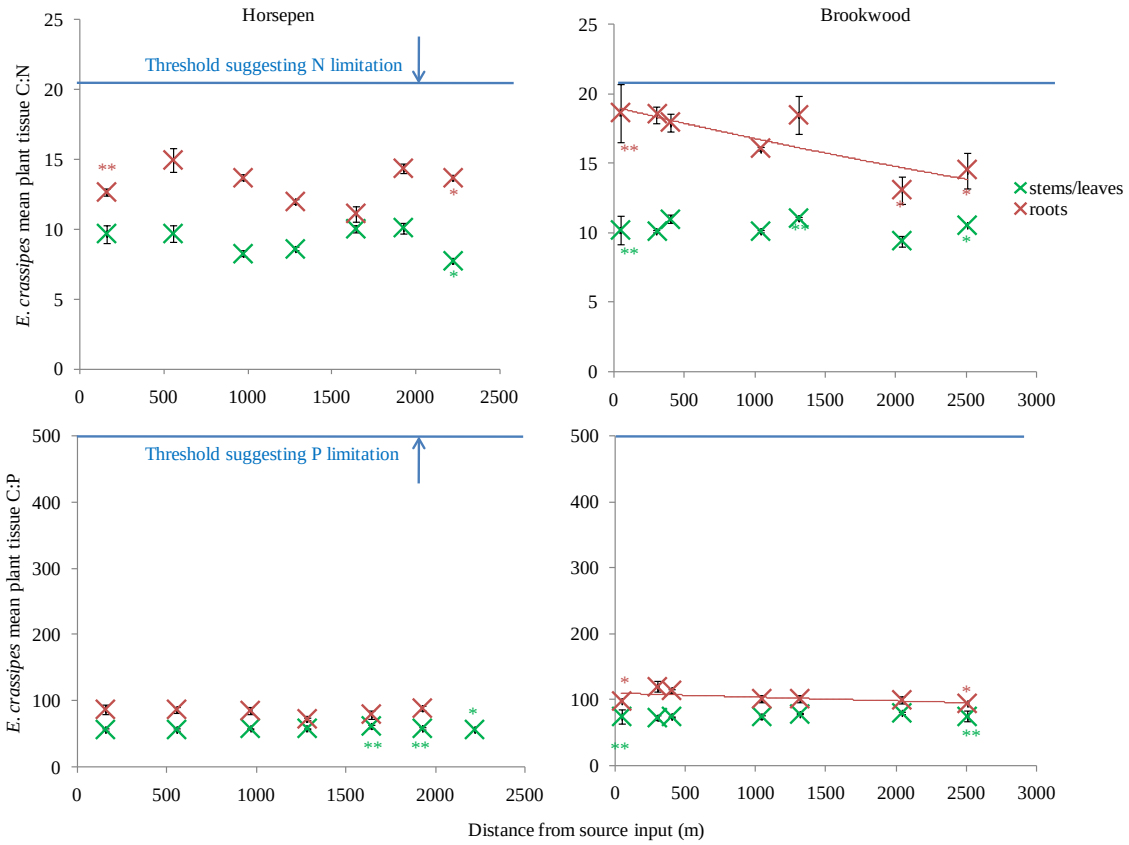

Fig. 15 *E. crassipes* mean emergent (stems/leaves) and submerged (roots) plant tissue C/N and C/P ratios measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 12 Least squares linear regression and exponential curve estimation between *E. crassipes* emergent (stems/leaves) and submerged (roots) plant tissue C/N and C/P ratios and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			Ratio	R^2	p
Horsepen	*E. crassipes*	stems/leaves	linear	C:N (19)	-0.058	0.939
				C:P (17)	0.040	0.216
			curve	C:N	-0.058	0.919
				C:P	0.043	0.211
		roots	linear	C:N (18)	-0.035	0.521
				C:P (18)	-0.040	0.567
			curve	C:N	-0.033	0.509
				C:P	-0.040	0.562
Brookwood	*E. crassipes*	stems/leaves	linear	C:N (17)	0.014	0.285
				C:P (19)	0.038	0.207
			curve	C:N	0.015	0.282
				C:P	0.041	0.200
		roots	linear	C:N (19)	0.451	0.001
				C:P (17)	0.261	0.021
			curve	C:N	0.470	0.001
				C:P	0.270	0.019

3.2.2 *E. crassipes* Relative Growth Rates and Nutrient Uptake Rates

Eichhornia crassipes relative growth rates (RGRs) for Horsepen decreased linearly (R^2=0.164; p=0.048) suggesting there is a decline in the nutrient removal potential for this species downstream from this tributary. RGRs for Brookwood were variable and did not change with increased distance from source input (Fig 16 and Table 13) suggesting there is more nutrient removal potential for *E. crassipes* along this tributary.

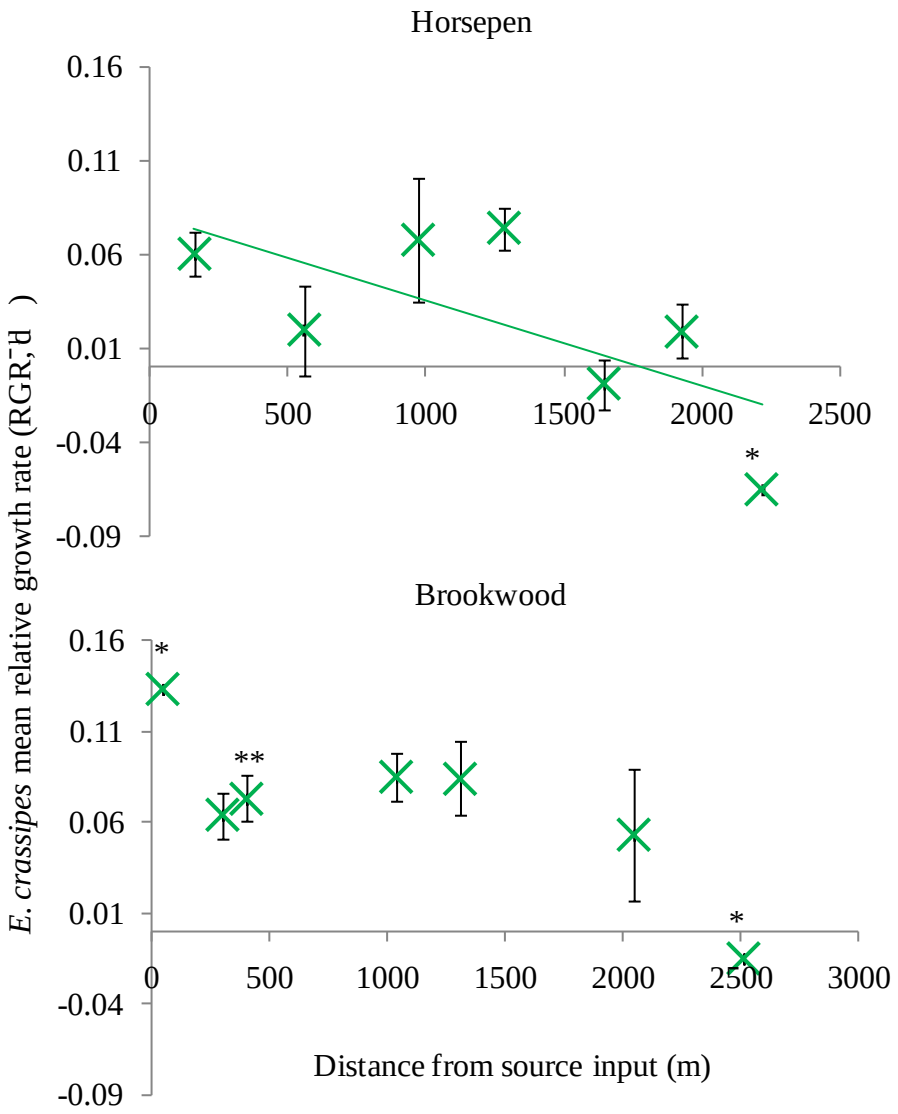

Fig. 16 *E. crassipes* mean relative growth rates (RGR, d^{-1}) measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

Table 13 Least squares linear regression and exponential curve estimation between *E. crassipes* relative growth rates and total nitrogen and phosphorus uptake rates and distance from nutrient source input. Red text indicates highest R^2 with confidence level of 95% corresponding to a significance level of $p<0.05$. The number of replicates is noted in parentheses.

Tributary	Plant Species			R^2	p
Horsepen	*E. crassipes*	relative growth rate (19)	linear	0.164	0.048
		g N d^{-1} (18)	linear	0.144	0.067
		g P d^{-1} (19)	linear	0.108	0.093
Brookwood	*E. crassipes*	relative growth rate (16)	linear	0.149	0.078
		g N d^{-1} (10)	linear	-0.019	0.388
			curve	-0.015	0.378
		g P d^{-1} (12)	linear	0.225	0.068
			curve	0.196	0.084

Nutrient uptake rates were low for both tributaries. Nitrogen uptake rates for Horsepen showed variability with increased distance from source input while phosphorus uptake rates appeared uniform across distance. Nutrient uptake rates for Brookwood appeared uniform across distance (Fig. 17). Brookwood showed higher mean nitrogen uptake rates across all stations compared to Horsepen (0.007±0.001 and 0.004±0.001 SE, respectively) while phosphorus uptakes were the same (0.001±0.0001 SE). Regression analysis did not show a trend between nutrient uptake rates and increased distance from source input (Table 13).

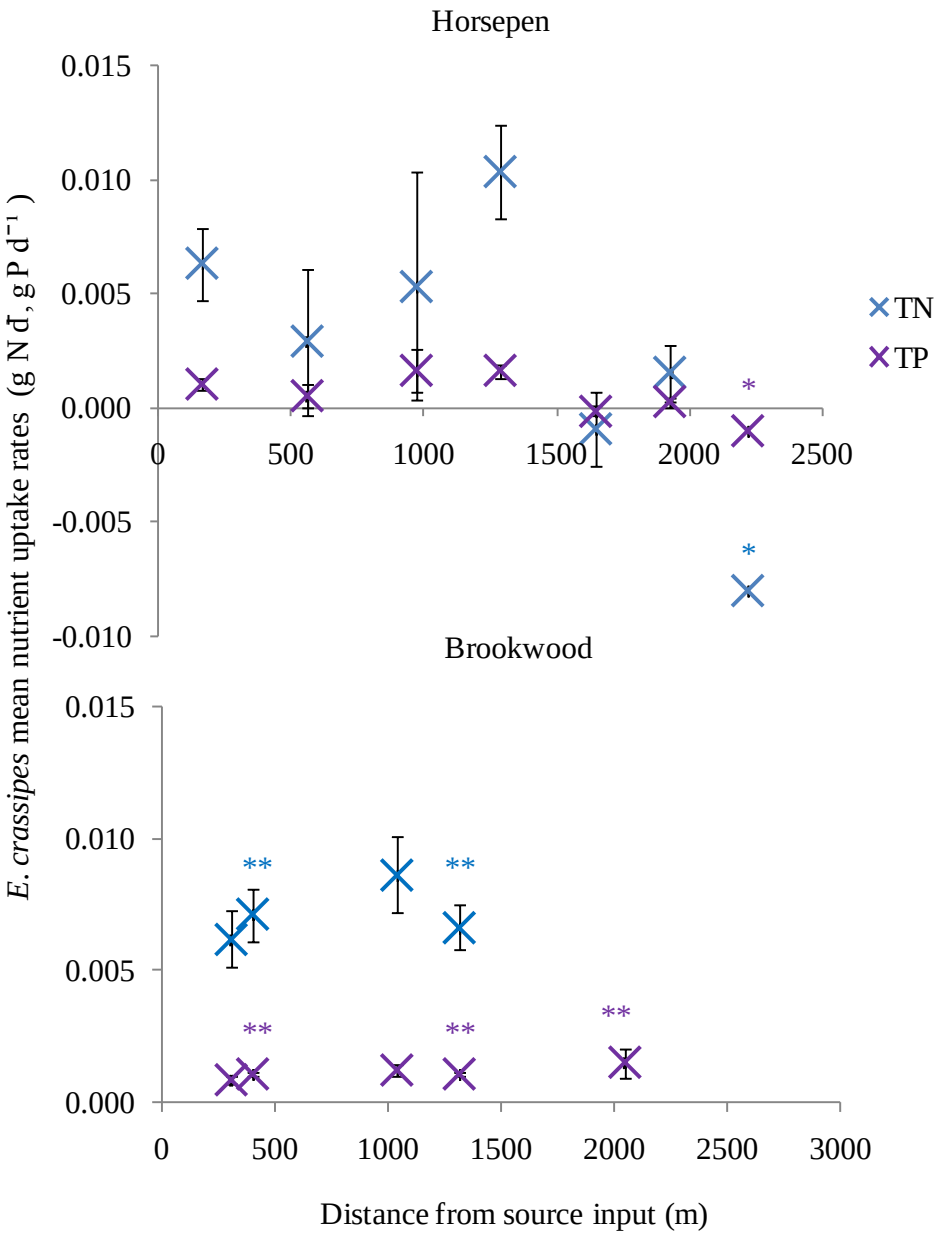

Fig. 17 *E. crassipes* mean nutrient uptake rates (g N d^{-1}, g P d^{-1}) measured from stations downstream from nutrient source input. Bars signify standard error (SE). **n*=1, ***n*=2, *n*=3.

4. DISCUSSION

4.1 Nutrient Storage Compartments in Armand Bayou

Contrary to my hypotheses, there were very few significant relationships between nutrient storage compartments (plant tissue and sediment nutrient content and pore water nutrient concentration) and increasing distance from nutrient source. Water column total nitrogen concentration showed the most significant [exponential] decrease, which was also paralleled in water column nitrate concentration, between stations HB7 and HB6 (closest to source input). This suggests a sink for this constituent immediately adjacent to the source of sewage effluent. However, the three species of aquatic vegetation sampled in the ABSS study showed no significant response to changes in water column nutrient concentration indicating these species are saturated and alone are not an effective nutrient sink. Saturated plants reach a threshold due to increased external nutrient loading in which they can no longer increase their nutrient uptake (Millar, 1955). Additionally, low mean stem density for exotics *P. stratiotes* (0.02±0.002 and 0.05±0.03 SE m^{-2}) across all stations and for *A. philoxeroides* (26±1 and 50±7 SE m^{-2}) across half the stations for Horsepen and Brookwood, respectively, confirm the lack of sufficient plant biomass to effectively improve water quality. In order to explain this nutrient sink between these two stations nearest to the sewage outfall the presence of a variety of other aquatic plant species that were not surveyed should be taken into account.

4.2 Water Column Nutrient Concentration

The use of wetlands for water quality improvement and wastewater impacts on wetland biota has been studied in various wetland treatment systems (Knight et al. 1993). In our study there was little evidence that nutrients in the water column were removed from the system via assimilation by exotic and restored, native vegetation, sediment adsorption or pore water retention. Although all water column nutrient concentrations for Horsepen and some nutrient concentrations for Brookwood (NO_2^- and NH_4^+) significantly decreased downstream, this largely appeared to be a dilution effect due to the increased volume of water downstream from sources of input. This downstream water source was from Mud Lake and is linked to Galveston Bay via Clear Lake. Although I did not measure the water quality in Mud Lake to confirm that it had better water quality than Horsepen Bayou, it is likely that the tidally-flushed Mud Lake has lower nutrient concentrations than Horsepen Bayou. We found that the wetland storage compartments investigated in Armand Bayou were not removing a sufficient amount of nutrients to effectively improve overall water quality downstream from sources of input. However, some studies have reported nutrient assimilation in treatment wetlands that improved overall surface water quality (Brantley et al. 2008; Blahnik and Day 2000). In a Louisiana freshwater forested wetland receiving wastewater effluent surface water showed on average significant reductions in nitrate, ammonium and phosphate concentrations downstream from source input which was attributed to longer residence times per unit area because of the decreased water velocities and larger surface area to water volume ratios characteristic to shallower depths (Blahnik and Day 2000). Brantley

et al. (2008) also found on average significant reductions in water column total nitrogen, nitrate, ammonium and phosphate concentrations in a Louisiana forested treatment wetland which were related to nutrient assimilation by trees as observed in the enhanced productivity from the fertilizing effects of the sewage discharge downstream. Mallin et al. (2002) found a decline in ammonium concentrations in a North Carolina detention basin which they attributed to plant uptake from a larger diverse plant population and a construction design of nutrient inputs at the upper end of the system. They also found increases in water column phosphate and nitrate concentrations in another detention basin, which they concluded was a result of additional source input from golf course runoff.

4.3 Sediment Nutrient Content and Pore Water Nutrient Concentration

Low sediment and pore water nutrients for Horsepen and Brookwood suggest these storage compartments are not providing effective sinks to improve water quality for either tributary. Alternatively, a subtropical marsh influenced by agricultural runoff proved to be an efficient nutrient sink where pore water ammonium and phosphate concentrations were 2.5 and 5.5, respectively, times higher than the water column as a result of high accretion rates and sediment nutrient remineralization (Soto-Jimenez et al. 2003). Unlike the system in Armand Bayou, external nutrient loading in the Florida Everglades resulted in high sediment total nitrogen (2.4-3.6%) which remained constant with increasing distance while total phosphorus decreased exponentially to an average of 0.05 % downstream from nutrient source (Vaithiynathan and Richardson 1997). The authors also reported elevated pore water phosphate (0.91 mg/L P) and ammonium (18

mg/L N) concentrations due to agricultural runoff (Vaithiyanathan and Richardson 1997). Daoust and Childers (2004) found similar results in sediment total nitrogen content (3.34 and 3.33% N) and total phosphorus content (0.03 and 0.04% P) between sites impacted by low level nutrient additions in the Everglades. In addition, the authors did not find any significant differences between pore water phosphate, ammonium, nitrate, or nitrite concentrations between sites.

4.4 Plant Tissue Nutrient Content

All three exotic plant species had a higher nutrient storage capacity in aboveground/emergent plant tissue than the native *S. californicus* for both tributaries. Belowground/submerged plant tissue total nitrogen for exotics also had higher nutrient storage capacity than native *S. californicus*. However, belowground/submerged plant tissue total phosphorus was similar for all species. *Schoenoplectus californicus* aboveground and belowground plant tissue nutrient content for Horsepen was within range for emergent plants (0.93 to 2.56% N and 0.14 to 0.30% P dry weight) in natural wetlands (Boyd 1978) and wetland plants exposed to increased nutrient loads (0.25 to 2.14% N and 0.13 to 1.07% P dry weight) (McJannet et al. 1995). Low plant tissue total nitrogen and significant decreases in total phosphorus with increasing distance from source input indicate this species is not providing a sink for nutrients that would aid in water quality improvement downstream. Aboveground/emergent plant tissue total nitrogen for exotics, *A. philoxeroides* and *P. stratiotes* were higher than emergent and floating, leaved plants in natural wetlands (1.86 to 3.79% N dry weight) (Boyd 1978) and nutrient enriched wetland plants (McJannet et al. 1995). This suggests some nutrient

removal potential for exotics. However, with no significant changes in aboveground/emergent nitrogen content over increasing distance from source input, these species are probably saturated and not providing an effective nitrogen sink to contribute to overall water quality improvement. Based on the bioassay results *Eichhornia crassipes* was the only exotic with higher emergent plant tissue total phosphorus for floating leaved plants in natural wetlands (0.14-0.40% P dry weight) (Boyd 1978) but within the range for nutrient enriched wetland plants (McJannet et al. 1995). This suggests a higher phosphorus storage capacity for this compartment, but with no change over increasing distance from nutrient source this compartment is probably saturated and not providing an effective phosphorus sink to contribute to overall water quality improvement. Belowground/submerged plant tissue total nitrogen and phosphorus content for *A. philoxeroides*, *P. stratiotes* and *E. crassipes* was within the range for emergent and floating, leaved plants in natural wetlands (Boyd 1978). However, significant increases in submerged plant tissue nutrient content with increasing distance from source input for *E. crassipes* and *P. stratiotes* suggest an additional source for water column nutrients.

The higher C/N ratios in aboveground and belowground plant tissue for native *S. californicus* for both tributaries suggest this species is limited in its requirements for nitrogen compared to the exotics. However, an enriched system such as Armand Bayou may not be nitrogen limited, but rather the higher carbon content may be explained by the contribution of structural carbohydrates (Sterner and Elser 2002). Mean aboveground and belowground plant tissue C/N ratio for *S. californicus* for both

tributaries was generally similar (<40) to a variety of emergent angiosperms surveyed in natural shallow lakes in Spain (C:N=36) which was attributed to the plants being nitrogen limited (Fernández-Aláez et al. 1999). Mean aboveground/emergent plant tissue C/N ratios for all three exotics were generally less than 10, which is much lower than the 27:1 ratio found in a variety of freshwater aquatic angiosperms (submerged and rooted with floating leaves) as well as the 36:1 ratio for emergent angiosperms (Fernández-Aláez et al. 1999). This suggests these exotics are better equipped to remove excess nitrogen for both tributaries. Similar results were observed for belowground/submerged plant tissue C/N ratios for exotics, *P. stratiotes* and *E. crassipes*. Mean belowground/submerged plant tissue C/N ratios were generally less than 15 and 20, respectively, which are much lower than the aquatic angiosperms surveyed. Conversely, mean belowground plant tissue C/N ratios for exotic *A. philoxeroides* were generally less than 25 which are much lower than the 36:1 ratio for emergent angiosperms surveyed (Fernández-Aláez et al. 1999). This suggests this storage compartment is not nitrogen limited, although this species was measured across only two stations for Horsepen (<800 m) and four stations for Brookwood (<400 m).

Mean aboveground and belowground plant tissue C/P ratios for native *S. californicus* were generally less than 300 for both tributaries even with a linear increase downstream for Brookwood. Mean aboveground plant tissue C/P ratios for exotic *A. philoxeroides* were even lower (<150) for both tributaries. This is much lower than the 500:1 C/P ratio for aquatic angiosperms (Duarte 1992) and the 790:1 C/P ratio for emergent angiosperms surveyed (Fernández-Aláez et al. 1999). Belowground plant

tissue C/P ratios for *A. philoxeroides* was measured at only one station for Horsepen and across four stations (<400 m) for Brookwood and were generally less than 200. Based on the phosphorus ratios for both aboveground and belowground storage compartments of this species it appears this emergent exotic is not phosphorus limited. Mean emergent plant tissue C/P ratios for exotic *P. stratiotes* were generally less than 150 and even lower (<80) for *E. crassipes* for both tributaries compared to the C/P ratio found in aquatic angiosperms (Duarte 1992) and the 704:1 C/P ratio found in floating leaved angiosperms (Fernández-Aláez et al. 1999) surveyed. The floating exotics *P. stratiotes* and *E. crassipes* appear to provide more nutrient removal potential compared to the emergent species for both tributaries based on the fact that they assimilate nutrients directly from the water column.

4.5 *E. crassipes* Relative Growth Rates and Nutrient Uptake Rates

Eichhornia crassipes, known to be a fairly productive species (Boyd 1971), has been shown to attain higher growth rates when influenced by secondarily treated sewage effluent compared to plants grown under normal conditions (Wooten and Dodd 1976). *Eichhornia crassipes* relative growth rates for Brookwood (0.082±0.007 d^{-1}) were higher compared to Horsepen (0.055±0.009 d^{-1}) yet both were higher compared to the same species in high nitrogen and phosphorus conditions as well as in natural wetlands (0.025 and 0.048 d^{-1}, respectively) (Henry -Silva et al. 2008; Hadad and Maine 2007). This suggests the nuisance potential of this species when influenced by high nutrient loads (Henry-Silva et al. 2008). Accelerated growth by *E. crassipes* under high nutrient conditions can lead to rapid colonization in aquatic systems subsequently affecting

biodiversity by outcompeting native species (Charudattan 2001). Additionally, dead biomass from large infestations can lead to increased rates of sedimentation and eutrophication and reduced water depth (Charudattan 2001).

Eichhornia crassipes mean total nitrogen and phosphorus uptake rates were low for Horsepen and Brookwood compared to uptake rates for this species found in constructed treatment wetlands (~0.548 g N m^{-2} d^{-1} and 0.096 g P m^{-2} d^{-1}) (Brix 1997). Higher assimilation rates (0.777 g N m^{-2} d^{-1} and 0.2 g P m^{-2} d^{-1}) during the summer for *E. crassipes* have been reported under nutrient enriched conditions due to its rapid growth rate and short hydraulic retention time which resulted from the shallow system used in the experiment (<1 m) (DeBusk et al. 1995). Reddy and DeBusk (1985) estimated potential nitrogen and phosphorus removal by *E. crassipes* over a 7 day detention period to be 1.28 and 0.24 g m^{-2} d^{-1}, respectively, during the summer months in nutrient enriched microcosm retention ponds. This is within the range of nutrient uptake rates (0.533 to 2.161 g N m^{-2} d^{-1}; 0.059 to 0.542 g P m^{-2} d^{-1}) found in aquaculture systems receiving additional ammonium and nitrate concentrations (Reddy and Tucker 1983). Reddy (1983) reported lower nitrogen uptake rates for *E. crassipes* (0.20 g N m^{-2} d^{-1}) due to low initial plant densities in wastewater retention basins. In this study low plant density (one specimen/enclosure) and retention time of only 7 days resulted in low nutrient uptake rates. Perhaps increasing both the number of individual plants deployed and retention time in Armand Bayou would yield higher and more comparable nutrient uptake rates for this species.

5. CONCLUSIONS

The results of this study indicate that dilution is the primary cause of decreasing water column nutrient concentrations with increasing distance from nutrient sources within Armand Bayou. The significant nitrogen sink observed between the two closest stations to the sewage outfall could not be determined in the ABSS study but may be accounted for by the additional aquatic plant species present. Low nutrient content in sediment and pore water suggest that these storage compartments are not effective sinks for improving water quality downstream. Restored, native *S. californicus* provided the largest plant biomass storage compartment within the system, but this species seems to have a limited ability to remove nitrogen from the water column. This suggests that in order to provide maximum nutrient uptake in systems with increased nutrient loads, this species would need to be restored in large quantities to increase the nutrient storage compartment for water quality improvement. Exotic species *A. philoxeroides*, *P. stratiotes* and *E. crassipes* have higher nutrient capacities, suggesting a potential for higher nutrient removal, although these particular species would also need to be in large quantities to provide enough biomass for effective nutrient removal and water quality improvement. Caution should be taken with this approach as this could lead to potential nuisance problems associated with invasive species infestations.

For resource managers, this study provides specific nutrient storage capacities of wetland compartments (e.g., aquatic wetland plants, sediment, and water column and pore water) influenced by anthropogenic impacts for evaluating water quality management plans. Additionally, this study helped to determine the fate of excess

nutrient loading into Armand Bayou. This understanding of where water column nutrients are transported and retained within the system can guide resource managers on the development of TMDLs within Armand Bayou watershed. The development of TMDLs will help improve standards of municipal waste discharge into wetlands. In wetland restoration practices where water quality improvement is a goal, this study provides nutrient retention capacities of specific wetland macrophytes ranging from restored, native emergent species to exotic floating species. Specific wetland plant nutrient removal potential can help resource managers decide on what species of aquatic vegetation to incorporate in removing excessive nutrient loads. In habitats where invasive, exotic species are abundant, this study suggests that those species are more efficient at nutrient uptake and lack of proper management of nutrient input will result in further proliferation of exotics. Improved management of invasive, exotic vegetation leads to lower costs associated with current eradication methods used on these species. The re-evaluation of standards of municipal waste discharge into wetlands by water quality managers would help to control nutrient uptake by invasive, exotic vegetation, subsequently limiting their proliferation.

REFERENCES

Adhikari A.R., K. Acharya, S.A. Shanahan, and X. Zhou. 2010. Removal of nutrients and metals by constructed and naturally created wetlands in the Las Vegas Valley, Nevada. *Environmental Monitoring and Assessment* 1-17. doi: 10.1007/s10661-010-1775-y.

Agami, M., and K.R. Reddy. 1990. Competition for space between *Eichhornia crassipes* (Mart.) Solms and *Pistia stratiotes* L. cultured in nutrient-enriched water. *Aquatic Botany* 38: 195-208.

Aoi, T., and T. Hayashi. 1996. Nutrient removal by water lettuce (*Pistia stratiotes*). *Water Science and Technology* 34 (7-8): 407-412.

Armstrong, F., R. Stearns, and J. Strickland. 1967. The measurement of upwelling and subsequent biological processes by means of the Technicon Autoanalyzer and associated equipment. *Deep Sea Research* 14 (3): 381-389.

Bernhardt, H., and A. Wilhelms. 1967. The continuous determination of low level iron, soluble phosphate and total phosphate with the AutoAnalyzer. Presented at the *Technicon Symposium on Automation in Analytical Chemistry*, 3 Oct, New York.

Blahnik, T., and J.W. Day. 2000. The effects of varied hydraulic and nutrient loading rates on water quality and hydrologic distributions in a natural forested wetland. *Wetlands* 20: 48-61.

Bouyoucos, G. 1962. Hydrometer method improved for making particle size analyses of soils. *Agronomy Journal* 54: 464.

Bowden, W.B. 1987. The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochemistry* 4 (3): 313-348.

Boyd, C.E. 1969. Production, mineral nutrient absorption, and biochemical assimilation by *Justicia Americana* and *Alternanthera philoxeroides*. *Archiv für Hydrobiologie* 66 (2): 139-160.

Boyd, C.E. 1970. Vascular aquatic plants for mineral nutrient removal from polluted waters. *Economic Botany* 24 (1): 95-103.

Boyd, C.E. 1971. Variation in the elemental content of *Eichhornia crassipes*. *Hydrobiologia* 38 (3-4): 409-414.

Boyd, C.E. 1978. Accumulation of dry matter, nitrogen and phosphorus by cultivated

water hyacinths: Chemical composition of wetland plants. In *Freshwater wetlands: Ecological processes and management potential*, eds. R.E. Good, D.F. Whigham, R.L. Simpson, 155-167. New York: Academic Press.

Boyt, F., S. Bayley, and J. Zoltek Jr. 1977. Removal of nutrients from treated municipal wastewater by wetland vegetation. *Journal Water Pollution Control Federation* 49 (5): 789-799.

Brantley, C.G., J.W. Day, R.R. Lane, E. Hyfield, J.N. Day, and J.-Y. Ko. 2008. Primary production, nutrient dynamics, and accretion of coastal freshwater forested wetland assimilation in Louisiana. *Ecological Engineering* 34: 7-22.

Brix, H. 1997. Do macrophytes play a role in constructed treatment wetlands? *Water Science and Technology* 35: 11-18.

Brix, H., and H.-H. Schierup. 1989. The use of aquatic macrophytes in water-pollution control. *Ambio* 18: 100-107.

Charudatten, R. 2001. Are we on top of aquatic weeds? Weed problems, control options, and challenges. Presented at an *International Symposium on World's Worst Weeds*, 12, Nov, Brighton, United Kingdom.

Cornwell, D.A., J. Zoltek Jr., C.D. Patrinely, T. deS. Furman, and J.I. Kim. 1977. Nutrient removal by water hyacinths. *Journal Water Pollution Control Federation* 49 (1): 57-65.

Daoust, R.J., and D.L. Childers. 2004. Ecological effects of low-level phosphorus additions on two plant communities in a neotropical freshwater wetland ecosystem. *Oecologia* 141: 672-686.

Davis, M., and D. Cornwell. 1991. *Introduction to environmental engineering*, 2nd edition. New York: McGraw-Hill, Inc.

Day, J.J.W., J.-Y. Ko, J. Rybczyk, D. Sabins, R. Bean, G. Berthelot, C. Brantley, L. Cardoch, W. Conner, J.N. Day, A.J. Englande, S. Feagley, E. Hyfield, R. Lane, J. Lindsey, J. Mistich, E. Reyes, and R. Twilley. 2004. The use of wetlands in the Mississippi Delta for wastewater assimilation: A review. *Ocean & Coastal Management* 47: 671-691.

DeBusk, T.A., J.E. Peterson, and K.R. Reddy. 1995. Use of aquatic and terrestrial plants for removing phosphorus from dairy wastewaters. *Ecological Engineering* 5: 371-390.

de Lange, P., R. Gardner, P. Champion, and C.C. Tanner. 1998. *Schoenoplectus*

californicus (Cyperaceae) in New Zealand. *New Zealand Journal of Botany* 36: 319-327.

Dhote, S., and S. Dixit. 2009. Water quality improvement through macrophytes: A review. *Environmental Monitoring and Assessment* 152: 149-153.

Diaz, R.J., and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. *Science* 321: 926-929.

Duarte, C.M. 1992. Nutrient concentration of aquatic plants: Patterns across species. *Limnology and Oceanography* 37: 882-889.

Dunigan, E., R. Phelan, and Z. Shamsuddin. 1975. Use of water hyacinths to remove nitrogen and phosphorus from eutrophic waters. *Hyacinth Control Journal* 13: 59-61.

Dunne, E.J., and K.R. Reddy. 2005. Phosphorus biogeochemistry of wetlands in agricultural watersheds. In *Nutrient management in agricultural wetlands: A wetlands solution*, eds. E.J. Dunne, K.R. Reddy and O.T. Carton, 105-119. The Netherlands: Wageningen Academic Publishers.

East, J., and J. Hogan. 2003. Hydrologic, water-quality, and biological data for three water bodies, Texas gulf coast plain, 2000-2002. Open File Report No. 03-459, Department of the Interior, US Geological Survey, Austin, Texas. 131 pp.

Elshorbagy, A., R. Teegavarapu, and L. Ormsbee. 2005. Total maximum daily load (TMDL) approach to surface water quality management: Concepts, issues, and applications. *Canadian Journal of Civil Engineering* 32: 442-448.

Faulkner, S.P., and C.J. Richardson. 1989. Physical and chemical characteristics of freshwater wetland soils. In *Constructed wetlands for wastewater treatment: Municipal, industrial and agricultural*, ed. D.A. Hammer, 41-72. Chelsea, Michigan: Lewis Publishers, Inc.

Fernández-Aláez, M., C. Fernández-Aláez, and E. Bécares. 1999. Nutrient concentration in macrophytes in Spanish shallow lakes. *Hydrobiologia* 408: 317-326.

Fourqurean, J., and J. Zieman. 1992. Phosphorus limitation of primary production in Florida Bay: Evidence from C: N: P ratios of the dominant seagrass *Thalassia testudinum*. *Limnology and Oceanography* 37: 162-171.

Fox, L., P. Struik, B. Appleton, and J. Rule. 2008. Nitrogen phytoremediation by water

hyacinth (*Eichhornia crassipes* (Mart.) Solms). *Water, Air, & Soil Pollution* 194: 199-207.

Geiger, S., J. Luzier, and J. Jackson. 1993. Nitrogen and phosphorus reduction in secondary effluent using a 15-acre, multiple-celled reed canarygrass (*Phalaris arundinacea*) wetland. In *Constructed wetlands for water quality improvement*, ed. G.A. Moshiri, 321-329. Boca Raton, Florida: CRC Press.

Gordon, A.S., W.J. Cooper, and D.J. Scheidt. 1986. Denitrification in marl and peat sediments in the Florida Everglades. *Applied and Environmental Microbiology* 52 (5): 987-991.

Hadad, H.R., and M.A. Maine. 2007. Phosphorus amount in floating and rooted macrophytes growing in wetlands from the Middle Paraná River floodplain (Argentina). *Ecological Engineering* 31: 251-258.

Harris County Flood Control District. 2010. Stormwater detention: How it works. Project Brays. http://www.projectbrays.org/stormwater.asp?flash=yes Accessed 7 August 2010.

Harwood J., and A. Kuhn. 1970. A colorimetric method for ammonia in natural waters. *Water Research* 4: 805-811.

Henry-Silva, G.G., A.F.M. Camargo, and M.M. Pezzato. 2008. Growth of free-floating aquatic macrophytes in different concentrations of nutrients. *Hydrobiologia* 610: 153-160.

Hoagland, C., L. Gentry, M. David, and D. Kovacic. 2001. Plant nutrient uptake and biomass accumulation in a constructed wetland. *Journal of Freshwater Ecology* 16: 527-540.

Howarth, R.W., A. Sharpley, and D. Walker. 2002. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. *Estuaries* 25: 656-676.

Imaoka, T., and S.Teranishi. 1988. Rates of nutrient uptake and growth of the water hyacinth *Eichhornia crassipes* (Mart.) Solms. *Water Research* 22 (8): 943-951.

Kadlec, R.H., and S.D. Wallace. 2009. *Treatment wetlands*, 2nd edition. Boca Raton, Florida: CRC Press.

Knight, R.L., R.W. Ruble, R.H. Kadlec, and S. Reed. 1993. Wetlands for wastewater treatment: Performance database. In *Constructed wetlands for water quality improvement*, ed. G.A. Moshiri, 35-58. Boca Raton, Florida: CRC Press.

Kramer, M. 2010. Informal conversation. Armand Bayou Nature Center. Pasadena, Texas.

Lawrence, A.I., J. Marsalek, J.B. Ellis, and B. Urbonas. 1997. Stormwater detention and BMPs. *Journal of Hydraulic Research* 34 (6): 799-813.

Mallin, M.A., S.H. Ensign, T.L. Wheeler, and D.B. Mayes. 2002. Pollutant removal efficacy of three wet detention ponds. *Journal of Environmental Quality* 31: 654-660.

Maddox, D.M., L.A. Andres, R.D. Hennessey, R.D. Blackburn, and N.R. Spencer. 1971. Insects to control alligatorweed: An invader of aquatic ecosystems in the United States. *Bioscience* 21 (19): 985-991.

McFarlane, R.W. 1991. An environmental inventory of the Armand Bayou Coastal Preserve. Final Report to Galveston Bay National Estuary Program, 66 pp.

McJannet, C.L., P.A. Keddy, and F.R. Pick. 1995. Nitrogen and phosphorus tissue concentrations in 41 wetland plants: A comparison across habitats and functional groups. *Functional Ecology* 9: 231-238.

Millar, C.E. 1955. *Soil fertility,* New York: John Wiley & Sons, Inc.

Mitchell, D.S., and N.M. Tur. 1975. The rate of growth of *Salvinia molesta* (S. Auriculata Auct.) in laboratory and natural conditions. *Journal of Applied Ecology* 12 (1): 213-225.

Mitsch, W.J., and J.G. Gosselink. 2007. *Wetlands*, 4th edition. New Jersey: John Wiley & Sons, Inc.

National Research Council (NRC). 1993. *Managing wastewater in coastal urban areas.* Washington, D.C.: National Academy Press.

Nichols, D. 1983. Capacity of natural wetlands to remove nutrients from wastewater. *Journal Water Pollution Control Federation* 55: 495-505.

Nixon, S.W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. *Ophelia* 41: 199-219.

Officer, C.B. 1979. Discussion of the behaviour of nonconservative dissolved constituents in estuaries. *Estuarine and Coastal Marine Science* 9: 91-94.

Reddy, K.R. 1983. Fate of nitrogen and phosphorus in a wastewater retention reservoir containing aquatic macrophytes. *Journal of Environmental Quality* 12 (1): 137-141.

Reddy, K.R., D.L. Sutton, and G. Bowes. 1983. Freshwater aquatic plant biomass production in Florida. *Soil and Crop Science Society of Florida* 42: 28-40.

Reddy, K.R., and J.C. Tucker. 1983. Productivity and nutrient uptake of water hyacinth, *Eichhornia crassipes* I. Effect of nitrogen source. *Economic Botany* 37 (2): 237-247.

Reddy, K.R., and W.F. DeBusk. 1985. Nutrient removal potential of selected aquatic macrophytes. *Journal of Environmental Quality* 14: 459-462.

Reddy, K.R., and W.F. DeBusk. 1987. Nutrient storage capabilities of aquatic and wetland plants. In *Aquatic plants for water treatment and resource recovery*, eds. K.R. Reddy and W.H. Smith, 337-357. Orlando, Florida: Magnolia Publishing Inc.

Richardson, C.J., and C.B. Craft. 1993. Effective phosphorus retention in wetlands: Fact or fiction? In *Constructed wetlands for water quality improvement*, ed. G.A. Moshiri, 271-282. Boca Raton, Florida: CRC Press.

Rogers, H.H., and D.E. Davis. 1972. Nutrient removal by water hyacinth. *Weed Science* 20 (5): 423-428.

Sakadevan, K., and H. Bavor. 1998. Phosphate adsorption characteristics of soils, slags and zeolite to be used as substrates in constructed wetland systems. *Water Research* 32: 393-399.

Sakadevan, K., and H. Bavor. 1999. Nutrient removal mechanisms in constructed wetlands and sustainable water management. *Water, Science & Technology* 40: 121-128.

Scarsbrook E., and D. Davis. 1971. Effect of sewage effluent on growth of five vascular aquatic species. *Hyacinth Control Journal* 9: 26-30.

Sheffield, C.W. 1967. Water hyacinth for nutrient removal. *Hyacinth Control Journal* 6: 27-30.

Shen, J., M. Shen, X. Wang, and Y. Lu. 2005. Effect of environmental factors on shoot emergence and vegetative growth of alligatorweed (*Alternanthera philoxeroides*). *Weed Science* 53: 471-478.

Smith, V.H. 2003. Eutrophication of freshwater and coastal marine ecosystems a global problem. *Environmental Science and Pollution Research* 10 (2): 126-139.

Sollie, S., H. Coops, and J. Verhoeven. 2008. Natural and constructed littoral zones as nutrient traps in eutrophicated shallow lakes. *Hydrobiologia* 605: 219-233.

Soto-Jimenez, M.F., F. Paez-Osuna, and H. Bojorquez-Leyva. 2003. Nutrient cycling at the sediment–water interface and in sediments at Chiricahueto marsh: A subtropical ecosystem associated with agricultural land uses. *Water Research* 37 (4): 719-728.

Sterner, R.W., and J.J. Elser. 2002. *Ecological stoichiometry: The biology of elements from molecules to the biosphere*, New Jersey: Princeton University Press.

Strickland J., and T. Parsons. 1972. *A practical handbook of seawater analysis*, 2nd edition. Ottawa: Fisheries Research Board of Canada.

Surrency D. 1993. Evaluation of aquatic plants for constructed wetlands. In *Constructed wetlands for water quality improvement*, ed. G.A. Moshiri, 349-386. Boca Raton, Florida: CRC Press.

Tanner, C.C. 1996. Plants for constructed wetland treatment systems: A comparison of the growth and nutrient uptake of eight emergent species. *Ecological Engineering* 7: 59-83.

Vaithiyanathan, P., and C.J. Richardson. 1997. Nutrient profiles in the Everglades: Examination along the eutrophication gradient. *Science of the Total Environment* 205: 81-95.

Valderrama, J.C. 1981. The simultaneous analysis of total nitrogen and total phosphorous in natural waters. *Marine Chemistry* 10: 109-122.

Walker, W.W.Jr. 1987. Phosphorus removal by urban runoff detention basins. *Lake and Reservoir Management* 3: 314-326.

Wong, T.H.F., P.F. Breen, and N.L.G. Somes. 1999. Ponds vs wetlands performance considerations in stormwater quality management. Presented at the *1st South Pacific Conference on Comprehensive Stormwater and Aquatic Ecosystem Management*, 22-26 Feb, Auckland, New Zealand.

Wooten, J.W., and J.D. Dodd. 1976. Growth of water hyacinths in treated sewage effluent. *Economic Botany* 30: 29-37.

Zimmels, Y., F. Kirzhner, and A. Malkovskaja. 2006. Application of *Eichhornia*

crassipes and *Pistia stratiotes* for treatment of urban sewage in Israel. *Journal of Environmental Management* 81: 420-428.

APPENDIX

ABSS TABLES

Table A-1 Mean water column nutrient concentrations (±SE) (mg/L N, P) measured from stations downstream from nutrient source input. *n*=3.

Tributary	Station	Distance (m)	Nutrient	Concentration (mg/L N, P)	±SE
Horsepen	HB7	159.94	TN	14.710	0.131
			NO_3^-	11.987	0.907
			NH_4^+	0.505	0.007
			NO_2^-	0.170	0.003
			PO_4^{3-}	3.288	0.482
	HB6	555.39	TN	3.575	0.127
			NO_3^-	2.072	0.076
			NH_4^+	0.363	0.025
			NO_2^-	0.145	0.004
			PO_4^{3-}	0.788	0.025
	HB5	970.08	TN	2.940	0.050
			NO_3^-	1.645	0.025
			NH_4^+	0.202	0.006
			NO_2^-	0.123	0.002
			PO_4^{3-}	1.127	0.006
	HB4	1282.57	TN	4.915	0.180
			NO_3^-	3.042	0.091
			NH_4^+	0.425	0.010
			NO_2^-	0.183	0.012
			PO_4^{3-}	1.157	0.007
	HB3	1641.99	TN	2.095	0.044
			NO_3^-	1.199	0.008
			NH_4^+	0.041	0.007
			NO_2^-	0.101	0.002
			PO_4^{3-}	0.821	0.010
	HB2	1926.54	TN	1.334	0.031
			NO_3^-	0.607	0.019
			NH_4^+	0.024	0.000
			NO_2^-	0.012	0.003
			PO_4^{3-}	0.722	0.013
	HB1	2218.88	TN	0.688	0.008
			NO_3^-	0.101	0.002
			NH_4^+	0.020	0.001
			NO_2^-	0.001	0.0001
			PO_4^{3-}	0.542	0.001

Tributary	Station	Distance (m)	Nutrient	Concentration (mg/L N, P)	±SE
Brookwood	BW8	44.51	TN	0.989	0.019
			NO_3^-	0.151	0.003
			NH_4^+	0.206	0.016
			NO_2^-	0.029	0.0002
			PO_4^{3-}	0.155	0.006
	BW7	181.75	TN	1.074	0.021
			NO_3^-	0.118	0.002
			NH_4^+	0.315	0.003
			NO_2^-	0.027	0.0002
			PO_4^{3-}	0.188	0.010
	BW6	303.33	TN	0.780	0.037
			NO_3^-	0.029	0.001
			NH_4^+	0.162	0.003
			NO_2^-	0.010	0.001
			PO_4^{3-}	0.280	0.017
	BW5	400.33	TN	0.641	0.028
			NO_3^-	0.034	0.002
			NH_4^+	0.103	0.006
			NO_2^-	0.011	0.001
			PO_4^{3-}	0.296	0.006
	BW4	1039.57	TN	1.122	0.063
			NO_3^-	0.316	0.003
			NH_4^+	0.116	0.029
			NO_2^-	0.024	0.004
			PO_4^{3-}	0.515	0.005
	BW3	1311.61	TN	0.604	0.025
			NO_3^-	0.030	0.010
			NH_4^+	0.009	0.001
			NO_2^-	0.004	0.001
			PO_4^{3-}	0.387	0.005
	BW2	2042.76	TN	1.116	0.022
			NO_3^-	0.398	0.018
			NH_4^+	0.011	0.002
			NO_2^-	0.003	0.001
			PO_4^{3-}	0.645	0.001
	BW1	2509.87	TN	0.875	0.030
			NO_3^-	0.295	0.022
			NH_4^+	0.018	0.005
			NO_2^-	0.007	0.003
			PO_4^{3-}	0.588	0.004

Table A-2 Mean plant total biomass (±SE) (kg dry weight/station) of exotics *A. philoxeroides* and *P. stratiotes* and restored, native *S. californicus* measured from stations downstream from nutrient source input. **n*=1, ***n*=2, *n*=3.

Tributary	Station	Distance (m)	Species	Plant total biomass (kg dry weight/station)	± SE
Horsepen	HB7	159.94	***A. philoxeroides*	8.56	0.83
	HB6	555.39	**S. californicus*	542.38	
			P. stratiotes	0.0005	0.0001
	HB5	970.08	*S. californicus*	364.50	87.56
			**A. philoxeroides*	0.46	
			P. stratiotes	0.0005	0.0001
	HB4	1282.57	***S. californicus*	180.00	56.75
			***P. stratiotes*	0.0004	0.00003
	HB3	1641.99	*S. californicus*	331.60	41.88
			P. stratiotes	0.0004	0.0001
	HB2	1926.54	***S. californicus*	631.44	30.08
			P. stratiotes	0.0006	0.0001
	HB1	2218.88	*P. stratiotes*	0.0005	0.0001

Tributary	Station	Distance (m)	Species	Plant total biomass (kg dry weight/station)	± SE
Brookwood	BW8	44.51	*A. philoxeroides*	39.50	8.10
	BW7	181.75	***A. philoxeroides*	76.96	25.52
			P. stratiotes	0.0008	0.0003
	BW6	303.33	*A. philoxeroides*	5.10	1.35
	BW5	400.33	***S. californicus*	328.83	91.73
			***A. philoxeroides*	12.15	0.87
			P. stratiotes	0.0003	0.0002
	BW4	1039.57	***S. californicus*	286.91	23.47
	BW3	1311.61	***S. californicus*	91.11	4.59
			***P. stratiotes*	0.0005	0.0002
	BW2	2042.76	*S. californicus*	164.53	6.07
	BW1	2509.87	***S. californicus*	301.05	26.61
			P. stratiotes	0.0001	0.00003

Table A-3 Mean stem density (±SE) (#/station) of exotics *A. philoxeroides* and *P. stratiotes* and restored, native *S. californicus* measured from stations downstream from nutrient source input. **n*=1, ***n*=2, *n*=3.

Tributary	Station	Distance (m)	Species	Stem density (#/station)	± SE
Horsepen	HB7	159.94	***A. philoxeroides*	23	3
	HB6	555.39	*S. californicus*	32	2
			A. philoxeroides	25	3
			P. stratiotes	0.01	0.00
	HB5	970.08	*S. californicus*	30	0
			***A. philoxeroides*	23	3
			P. stratiotes	0.02	0.00
	HB4	1282.57	*S. californicus*	30	3
			A. philoxeroides	20	3
			***P. stratiotes*	0.04	0.00
	HB3	1641.99	**S. californicus*	30	
			P. stratiotes	0.03	0.00
	HB2	1926.54	*S. californicus*	32	4
			P. stratiotes	0.03	0.00
	HB1	2218.88	*S. californicus*	28	6
			P. stratiotes	0.03	0.00

Tributary	Station	Distance (m)	Species	Stem density (#/station)	± SE
Brookwood	BW8	44.51	*A. philoxeroides*	80	15
	BW7	181.75	*A. philoxeroides*	58	4
			P. stratiotes	0.09	0.00
	BW6	303.33	*A. philoxeroides*	30	0
	BW5	400.33	*S. californicus*	42	4
			A. philoxeroides	32	9
			P. stratiotes	0.02	0.00
	BW4	1039.57	*S. californicus*	48	9
	BW3	1311.61	*S. californicus*	37	9
			***P. stratiotes*	0.04	0.00
	BW2	2042.76	*S. californicus*	35	5
	BW1	2509.87	*S. californicus*	35	3
			P. stratiotes	0.04	0.00

Table A-4 Pore water nutrient concentrations (mg/L N, P) measured from stations downstream from nutrient source input. *n*=1.

Tributary	Station	Distance (m)	Nutrient	Concentration (mg/L N, P)
Horsepen	HB7	159.94	TN	4.333
			NO_3^-	1.366
			NH_4^+	1.571
			NO_2^-	0.108
			PO_4^{3}	0.010
	HB6	555.39	TN	6.005
			NO_3^-	0.099
			NH_4^+	5.184
			NO_2^-	0.012
			PO_4^{3}	0.147
	HB5	970.08	TN	6.691
			NO_3^-	0.138
			NH_4^+	1.488
			NO_2^-	0.018
			PO_4^{3}	0.738
	HB4	1282.57	TN	7.403
			NO_3^-	0.010
			NH_4^+	6.569
			NO_2^-	0.006
			PO_4^{3}	0.134
	HB3	1641.99	TN	12.053
			NO_3^-	0.183
			NH_4^+	9.158
			NO_2^-	0.009
			PO_4^{3}	1.819
	HB2	1926.54	TN	6.163
			NO_3^-	0.040
			NH_4^+	5.209
			NO_2^-	0.004
			PO_4^{3}	1.329
	HB1	2218.88	TN	3.572
			NO_3^-	0.013
			NH_4^+	2.260
			NO_2^-	0.003
			PO_4^{3}	0.549

Tributary	Station	Distance (m)	Nutrient	Concentration (mg/L N, P)
Brookwood	BW8	44.51	TN	3.341
			NO_3^-	0.010
			NH_4^+	2.969
			NO_2^-	0.003
			PO_4^{3}	0.010
	BW7	181.75	TN	8.758
			NO_3^-	0.003
			NH_4^+	9.271
			NO_2^-	0.001
			PO_4^{3}	0.122
	BW6	303.33	TN	7.011
			NO_3^-	0.010
			NH_4^+	6.020
			NO_2^-	0.002
			PO_4^{3}	0.067
	BW5	400.33	TN	3.324
			NO_3^-	0.004
			NH_4^+	3.421
			NO_2^-	0.002
			PO_4^{3}	0.226
	BW4	1039.57	TN	5.249
			NO_3^-	0.003
			NH_4^+	4.177
			NO_2^-	0.004
			PO_4^{3}	0.622
	BW3	1311.61	TN	7.186
			NO_3^-	0.003
			NH_4^+	5.926
			NO_2^-	0.004
			PO_4^{3}	0.727
	BW2	2042.76	TN	2.872
			NO_3^-	0.007
			NH_4^+	1.653
			NO_2^-	0.002
			PO_4^{3}	0.121
	BW1	2509.87	TN	3.395
			NO_3^-	0.011
			NH_4^+	3.208
			NO_2^-	0.004
			PO_4^{3}	0.508

VITA

Allison Parnell received her Bachelor of Science degree in biology from Arkansas Tech University in 1998. She entered the Marine Resources Management program at Texas A&M University at Galveston in January 2007 and received her Master of Marine Resources degree in May 2011. Her research interests include nutrient distribution in water column, sediment, and exotic and restored native aquatic vegetation, and management of natural and restored wetland communities impacted by anthropogenic input.

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